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Article

Nutrient management in support of environmental and agricultural sustainability

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Abstract:

Given that we must farm land in order to eat, the total environmental burden imposed by farming a crop such as winter wheat in the UK appears to as close to the minimum as is possible when production is at its most efficient. The value of the services other than food production, such as flood water buffering, pollination, carbon storage and so on, that land can provide is relatively large compared with the value in reducing environmental burdens from pesticide use, nutrient pollution and greenhouse gas emissions by farming less intensively. More land will need to be brought into cultivation in order to provide the same amount of food if the intensity of farming is reduced but the loss of ecosystem services outweighs the reduction in. Nevertheless, losses of nutrients, especially nitrogen (N), from agriculture are a serious concern and the current cost of the environmental footprint of agriculture is significant compared with the value of the food it produces. This article examines nutrient burdens and analyses the means by which the total environmental burden might be reduced relative to productivity. These include increasing the efficiency of farming, removing constraints to yield, and establishing multiple uses for land at the same time as farming. It concludes that agronomic measures which improve nutrient capture and which obtain more yield per unit area are valuable means to avoid degradation of environmental quality because both nutrient pollution and land consumption can be avoided. Means that appear to offer a reduction of 5-10% in each are suggested.

Keywords: Soil, Modelling, Nutrients, Nutrient management, Nitrogen use efficiency, Crop production

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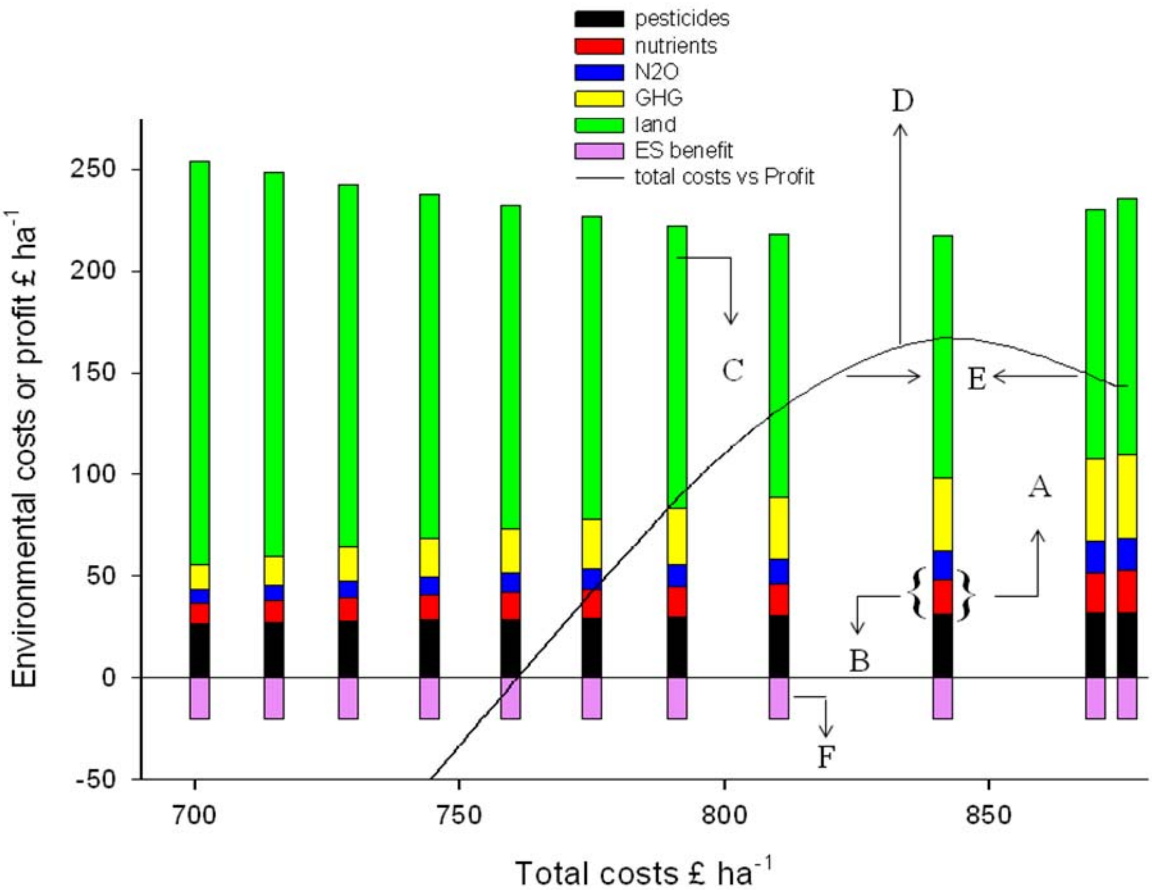
1. Introduction

Glendining *et al*. (2009) developed a framework for comparing the full profit and expenditure of agriculture: that is a framework that includes both the orthodox economic evaluation but also the environmental costs of agriculture. Essentially they considered the ratio of the value of all outputs to the costs of all inputs. Such a ratio is known as the Total Factor Productivity (TFP, Ehui *et al*., 1992). Glendining *et al*. (2009) calculated TFP for farm production but also included multiple outputs (grain and straw for *e.g.* bedding or fuel) as well as the environmental costs without which TFP would be biased (Barnett *et al*., 1994). They showed that for arable crops at least, current farming practice that is optimal for profit is very nearly optimal for environmental quality. Farming in Western Europe is very intensive but Glendining *et al*.'s (2009) conclusion was that reducing this intensity made little sense from an environmental point of view, if more land has to be brought into production in order to produce the same amount of food. Although reducing the intensity of farming did indeed appear to reduce nutrient and greenhouse gas emissions, the value of these gains was strongly outweighed by the value of the loss of the ecosystem services (ES) from the new land brought under the plough. Such argument support the concept of land sharing (Green et al., 2005)

If the argument is taken as sound that land consumption is the primary environmental burden to be minimised, the question then becomes: can current nutrient use in farming practice be improved in the context of this primary constraint? This article takes a systems approach to agriculture with a particular focus on minimising nutrient losses while avoiding the need to farm a greater area of land. Taking data published by Glendining *et al*. (2009) and analysing the profit and environmental costs more fully that are associated with producing a winter wheat crop as an example, we examine the scientific literature for ways in which farming might be improved to (i) deliver more profit without additional damage to the environment, (ii) deliver increased environmental quality without loss of yield, or (iii) increase both profit and environmental quality.

2. Results and Discussion

Figure 1: Environmental costs and profit as a result of growing winter wheat, plotted against total costs (orthodox economic costs plus environmental costs). Coloured bars: breakdown of environmental costs (pesticide burden, N and P loss, N_2O emission, other GHG emissions, loss of ES of native land and gain for arable land) Costs are driven mainly by increases in nitrogen fertiliser, but other costs vary with production too. Solid line: farmer's profit as a result of growing winter wheat at the same costs of production. The yield equivalent of this profit (same solid line) is indicated with reference to the scale on the right hand axis.



In Figure 1, the environmental costs of farming wheat are broken down into pesticides, N and P eutrophication, N_2O, other GHG emissions and loss of ecosystem service from land (bars) and displayed against the total cost of farming at increasingly intensive rates of production. The ecosystem service benefit of arable land is given as a negative cost (small pink bar) Also shown in Figure 1 is the response (line) conceived as profit plotted against the same intensity of production and this line displays the usual diminishing return to inputs. Yield is indicated on a secondary axis. Intensity of production is largely equivalent to the nitrogen applied in this example. Maximum profit is delivered at about the same point of the total input cost (abscissa) as the minimum environmental burden. Note, however, that the environmental cost is large at this minimum and far from trivial. Farming has a cost to the environment yet the fact that this environmental cost registers as larger than the farm profit is misleading, because environmental costs are not included in farm profit and loss accounts. If they were, the cost of food would change markedly.

2.1 Analysis of nutrient loss in relation to soil and environmental quality

To illustrate Glendining *et al*'s (2009) claim that reducing inputs to agriculture substantially does not improve environmental quality, consider the effect of reducing inputs to winter wheat from those which produce maximum yield to those which produce a zero financial return. The environmental cost associated with bringing the extra land into production that is needed to make up the shortfall in productivity on a national basis is £40 (Figure 1, change in height of green bars and expressed on a per hectare basis). Compared with this, the saving in all other environmental costs for the same change is £25. Of this £25 just £4 comes from reducing leaching N and P and less than £5 comes from the reduction in N_2O emissions (Figure 1).

If this is so, and if we accept that such use of land by farming is necessary to feed ourselves, then what insights does this full environmental economic analysis provide that can help to improve soil and environmental quality and especially reduce nutrient pollution? Possible means to achieve improved environmental quality are listed in the first column of Table 1 as well as indicated on Figure 1. **A**: The crop takes up and removes more nutrients from soil. **B**: Losses are reduced by whatever means. **C**: The efficiency of farming is improved. **D**: Constraints to crop growth are removed, which improves nutrient capture from soil. **E**: Dissemination and uptake of advice is improved to achieve optimal nutrient use and yield more widely. **F**: The value and extent of ES delivered by arable land is increased. This paper contains a summary of interventions that might achieve more efficient use of nitrogen and management of the crop as indicated in Figure 1. Fuller details can be found in the supplementary material online.

2.2 Specific means to improve soil and environmental quality

If unfarmed land supplies more ES than farmed, it makes sense to use the farmed area as efficiently as possible in order to avoid bringing land into production. So a loss of environmental quality can be avoided by increasing nutrient capture or by increasing yield, if this avoids consumption of land. Below we consider the ways in which environmental quality might be improved in this context.

2.2.1 Increase nutrient capture, A. Bradley *et al*. (2005) report the distribution of soil carbon in England and Wales. They give data from which the average density of carbon in the subsoil (30-70 cm) can be deduced as 0.5% C by weight. A typical C:N ratio for soil is 10 and so this implies very crudely that the average density of N in the subsoil is 0.05%, or half the threshold that Sylvester-Bradley *et al*. (2001) suggest might contribute significant amounts of N through mineralisation to the soil nitrogen supply to crops. Without a reported distribution or other measure of the variation it is unwise to speculate widely on the prevalence of soils containing more than 0.1% N in the UK but it is unlikely to be greater than 25% of all soils. On the other hand a soil containing 0.05% N in the soil volume from 30 to 100 cm may supply up to a third of the N contained in the topsoil and so about 20 or 30 kg ha^{-1} yr^{-1}, which would probably be leached if not captured by a crop. Capturing this could reduce nitrogen application by the equivalent amount.

2.2.2 Reduce losses B. Recent research in New Zealand and Australia has shown that Nitrification Inhibitors (NIs) can be extremely effective at reducing N_2O emissions from intensively grazed pasture (e.g. de Klein and Eckard 2008). If the nitrification of ammonium N is prevented, less leaching results and less N_2O is emitted during both nitrification and the subsequent denitrification of the resultant nitrate (NO_3). In summary, NIs applied to urine patches have been found to reduce N_2O emissions by 61-91%, to reduce NO_3 leaching by 30-79% and increase annual pasture yield by 0-36% (Di and Cameron 2002; 2003; 2005; 2008; Di et al. 2007). Rates of application were of the order of 10 kg DCD ha^{-1} at a cost of about £26 ha^{-1}.

Ammonia emissions can be reduced by the use of urease inhibitors (Dampney et al., 2003) and as a consequence indirect emissions of N_2O. There is some evidence to suggest that urea is more suited for use on soils prone to water-logging, whilst ammonium nitrate is more suited to less wet soils (Pathak and Nedwell, 2001). Although nitrification inhibitors were effective at field capacity in this study, they were ineffective under waterlogged conditions.

There is evidence that N_2O production increases at a non-linear rate when soil NO_3^- content exceeds crop demand. This has been reported under maize (Chantigny et al. 1998; McSwiney and Robertson 2005) and spring barley (Zebarth et al. 2008a). Under these conditions, and when conditions are also conducive to denitrification, splitting fertilizer N applications could reduce N_2O losses. Data presented by Zebarth et al. (2008a) suggest that lack of crop N uptake accounts for the relative increase in the proportion of mineral N that is emitted as N_2O at high rates of N application. Dalal et al. (2003) state that NO_3^- usually inhibits full denitrification to N_2, increasing the N_2O:N_2 ratio. This lends weight to the view that splitting applications should reduce losses and that correct N fertiliser guidance is imperative.

2.2.3 Decrease area of farmed land directly, C. If efficiencies can be found throughout the system, less land need be used, which reduces the environmental footprint of farming as a whole. Alternatively farmed land might be made more productive. Bulson *et al*. (1997) found a 29% increase in yields from barley and beans by intercropping at 150% total density compared with sole crops. Not only does this increase diversity (see F below) but less land is needed to produce the same amount of food, thus potentially increasing the supply of valuable ES on land no longer in arable production.

2.2.4 Remove constraints D. Removing constraints on productivity is likely to increase yield profitability; it may well affect the shape of the response curve shifting the optimum to either less or to more inputs. Careful use of growth regulators and fungicides with growth regulation properties, and maintaining the correct amount of N in soil for sufficient but not excessive growth, have been found to increase oilseed rape yields by 0.36 t ha^{-1} (Berry and Spink 2009). Lynch (2007) asserts that crops tend to develop roots in the surface soil for nutrient acquisition (particularly P) and in the subsoil for water. Ho *et al*. (2005) suggest that plants which increase the density of their roots in the topsoil are able to acquire P more effectively than those which do not. They argue further that genotypes with roots that proliferate deep in the soil are better able to withstand drought stress but also found dimorphic varieties that are able to adapt to either nutrient or water stress as appropriate. Waines and Ehdaie (2007) conclude that the root systems of modern wheat cultivars are small, having perhaps two thirds of the root mass of the landraces from which they derive. Whitmore et al. (2011) argue that the physical impediment to root growth is a more significant stress than the lack of accessibility of water. There thus seems scope for breeding to improve nutrient acquisition and to reduce other constraints to maximum yield and uptake such as water stress

Table 1: Means to improve soil and environmental quality

	Means	**Effect on soil and/or Environmental Quality**	**Driver**	**Possible means to achieve this**
A	Increase nutrient capture	Pollution is reduced if nutrient uptake is increased	Economics	1. use of subsoil N
B	Reduce losses of N to the environment	Lower emissions, reduces losses of N and/or P	Regulation	1. Nitrification Inhibitors 2, Splitting N application to reduce risk of loss
C	Decrease area of farmed land directly	Better provision of Ecosystem Services other than food and fibre production	Pressure from other users of land	1. Maximise efficiency of production
D	Remove constraints on yield	Less land needed, less N needed	Economics, yield increases if constraints are removed	1.Canopy management 2. Breeding 3. Management of root environment
E	Improve effectiveness of extension	Less waste, production can be optimised for both yield and environmental quality if extension is improved	Economics/Regulation	1. Improved guidance on use of nutrients 2. Improved understanding of N cycles 3. Precision management
F	Increase ES or functionality of farmland	Environmental quality and the wider public good can be increased by multi-functional land-use that enhances the services that arable land delivers	Pressure from other users of land	1. Increase organic matter levels 2. Adopt some form of mixed cropping

If water or root condition is limiting yield (Whitmore and Whalley, 2009), removing the constraint(s) and allowing the crop to yield to its full potential should improve nutrient use and benefit environmental quality by reducing losses and emissions. A further benefit of removing constraints to yield is that the farmer gains profit and so there is a strong economic pull from this direction, provided the action needed to remove the constraint is not excessively costly compared with the gain in yield. Because the costs of environmental pollution generally fall over the wider environment, measures to reduce nutrient loss have often been delivered through support (e.g. the EU single farm payment).

2.2.5 Improve effectiveness of extension, E. The benefit of improved extension can also be seen to act by moving the system towards the optimum production level (Figure 1, E). Removing constraints increases profit and increases the ratio of profit to environmental cost. If the economic and environmental optima are not precisely aligned (Glendining *et al*., 2009), advice to minimise deterioration of environmental quality will need to be focused on the environmental rather than economic optimum. Dailey *et al.* (2006) suggest that improved weather forecasts will make small (6 or 7 kg N ha^{-1}) but systematic improvements to N offtake and yield where N applications are constrained by regulation such as in Nitrate Vulnerable Zones. Recent research in NW Europe on the effectiveness of extension has tended to focus on meeting air and water quality standards (e.g Neeteson, 2000) but the advice itself is often based on achieving the economic optimum (e.g. Defra, 2010). Goulding et al. (2008) report that nitrogen use efficiency can be 60-90% on experimental plots but sometimes as little as 20-50% on cereal farms.

If N rates are applied at the rate to obtain the maximum or optimum yield given the existence of spatial variability (Whitmore and Van Noordwijk 1995), reduction of N use should reduce losses, but yields cannot be expected to improve detectably. There may be factors other than N supply that limit yield. If so, the simplistic approach of applying more N where growth is poor may be ineffective in increasing yield and risk environmental losses of N. It does seem likely, however, that N_2O emission and NO_3 leaching can be reduced by applying N in a spatially variable way, since several studies have found no loss of yield with a reduction in total N applied (e.g. Lark and Wheeler, 2003; Desbourdes *et al*. 2008; Robertson 2008). Savings in these terms have rarely been identified in the literature but could be substantial in global or national terms if most emissions derive from the under-use of applied N (perhaps 1-2 kg N_2O-N ha^{-1} or 50 kg NO_3-N ha^{-1} y^{-1}

2.2.6 Increase ES and functionality of farmed land, F. Even arable land provides ecosystem services. If a way could be found to increase the services or value of the services provided by arable land then the net environmental cost of farming would be less. For example, Whitmore and Schröder (2007) showed that intercropping could reduce nutrient use and nitrate leaching by 20 – 40 kg N ha^{-1}, without loss of yield or profit. Some measures have multiple effects. For example, increasing organic matter levels can increase soil fertility and, provided increased N supply from soil is taken into account, applications of fertiliser N can be reduced (Whitmore and Schröder, 1996). In addition, Watts *et al*. (2006) found that increasing organic carbon in soil by 0.1% could reduce the specific plough draught by 5 kPa, which represents a fuel saving of more than 5%. Arable land that helped control flooding could make a large and valuable contribution to both the full economy of the UK (Rouquette *et al*., 2009) and to environmental quality downstream. Clearly not all land can perform all services simultaneously, but UK arable production managed as a whole might achieve these things. Multi-functional land-use at the landscape scale is therefore a means to improve environmental quality as a whole while continuing to farm the food we need to feed ourselves. However, it is important to consider the tradeoffs between different measures to ensure that increasing carbon sequestration does not inadvertently lead to unexpected increases in e.g. N_2O emissions (Powlson et al., in press).

3. Methodology

Glendining *et al*. (2009) published methodology and spreadsheets for calculating the costs of producing and managing crops. Orthodox economic costs of inputs and operations were derived from agricultural almanacs such as Nix (2005) andABC (2005). Environmental costs were taken from national surveys of environmental impacts and Life Cycle Assessment (Atkinson *et al*., 2004; Pretty *et al*., 2000 & 2003; Hartridge and Pearce, 2001; Williams *et al*., 2006) and attributed to sectors on a land-area basis. Prices were correct at the time (2006) this study was carried out. Agricultural commodity prices have risen and fallen sharply in the intervening period, emphasising the caution that must be applied when attributing and comparing costs to such diverse quantities as the additional health care as a result of using pesticides, the loss of value of amenity water as a result of eutrophication, the potential damage to the economy following emissions of green-house gases (GHG), the value of wheat grain and so on.

We consider nutrient losses in particular here. Glendining *et al*. (2009) included nitrous oxide (N_2O) emissions as part of the full GHG burden of agriculture but here we have separated out the component of the GHG burden that can be attributed to N_2O emission from land when growing wheat (Figure 1, blue bars). Any N_2O emitted as a result of off-farm activity thus remains in the GHG data presented below (Figure 1, yellow bars), but emissions from soil under agriculture are displayed explicitly. Nutrient burdens refer to nitrate and phosphorus (P) losses by all means to surface water; ammonia is neglected in this data for a winter wheat crop. A large part of the environmental burden associated with P or N occurs from their joint impact on water quality. It is very difficult therefore to disentangle these effects and attribute them separately.

A major burden considered by Glendining *et al*. (2009) was the cost of bringing land that bears woodland or native vegetation into cropping. Woodland and land in its natural state provides far more extensive and valuable ES than cropped land and this loss contributes strongly to any financial balance sheet that includes the valuation of environmental factors. Natural land supports a wider diversity of organisms, it maintains an extensive network of pores in soil that can store or transmit water, and natural vegetation, especially forest, is good at removing pollutants from the atmosphere. Agricultural grassland is considerably more valuable than arable land in this context but less valuable than native vegetation. Values of ES were taken from Costanza *et al*. (1997; after discounting food and fibre production these were: arable land, US$38 ha^{-1}; woodland US$227 ha^{-1}) with the caveat that these numbers are as subject to change as any other in the economic calculations, and that services will become more valuable than we now assume if the land able to provide them becomes scarce (Farber *et al*., 2002). The source of additional land is assumed to be woodland, because grassland is already in agricultural use. If grassland were substituted instead, the conclusions would change little because the value of the ecosystem services provided by grassland was estimated at $165 ha^{-1}.

4. Conclusions

A degree of consumption of environmental services must be accepted if we are to feed ourselves; naturally this impacts on soil and environmental quality. Agronomic measures to save between 5 and 40 kg N ha^{-1} have been identified which in themselves might be valuable savings in fertiliser but if this N is wasted, its loss is a cost to environmental quality. More subtly, if these savings translate into extra yield, they represent up to 1 t ha^{-1}. UK winter wheat crops yield an average of 8 t ha^{-1} currently from just over 2M ha. An average increase to the bottom half of the distribution of 1 t ha^{-1} is equivalent to 0.1M ha of land from which ES might potentially be increased. Such a change would need to be made permanently, however, since native land is easy to consume but difficult to restore. The separate measures to improve soil or environmental quality proposed here are not mutually exclusive and indeed can be synergistic; for the most part a decrease in the environmental footprint at this level of production can be coupled with an increase in productivity and profit.

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Conflict of Interest

The authors declare no conflicts of interest.

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Supplementary material:
Nitrogen management in support of environmental and agricultural sustainability

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1. Executive Summary

The essential remit of the review is at first sight large: to improve understanding of the N cycle in agricultural systems as a basis for improving crop and environmental management. However it is impractical to review such a wide ranging subject in a shor publication and three key areas of interest were identified: (i) strategic nitrous oxide reduction – improved mineral fertiliser N timing strategies, (ii) reducing nitrogen losses from farms – UK potential for nitrification inhibitors and (iii) reduced tillage and N_2O emission

Nitrification inhibitors (NIs) have been found to work well in conditions where most benefit is to be expected: when applied to urine patches, for example. They have not been comprehensively evaluated under UK conditions, nor has a full life-cycle assessment (LCA) in the use of NIs been carried out to judge whether the use of these compounds represents a benefit or not. The apparent gains could be considerable, but comprehensive trials under controlled conditions in the laboratory and the field are needed to ensure that these benefits can be realised widely. There is a need to collect simple data for an LCA such as the manufacturing costs of the different NIs and to perform a full LCA on their deployment within UK farming systems. It is conceivable that some inhibitors will fare better than others in a full economic, environmental and life-cycle assessment but this simple evaluation has not been carried out, nor have all inhibitors been assessed under similar conditions. Information is piecemeal and uncoordinated

Reduced tillage more often than not leads to an increase in N_2O emissions. The increases by and large are small in absolute terms but can lead to a doubling or more of the emissions. Poorly-drained soils are particularly prone to substantial increases in N_2O emission on converting to minimum tillage especially if they become denser or compacted. Our knowledge of the soils affected and the conditions leading to enhanced N_2O emission are poor in the UK. There has been much international research in the area surveyed in the review, but information in the UK derives from few soils only, making it hard to recommend changes in UK practice. In particular rotational ploughing is more common in the UK than in North America and it is not clear to what extent this practice may alleviate the increased emissions of N_2O found in soils where reduced tillage is practised continuously.

Notwithstanding our lack of knowledge of the circumstances leading to emission of N_2O in this country and especially under reduced tillage, there appear to be fairly simple practices that the industry could adopt on wetter land that is more susceptible to N_2O emission. Splitting fertiliser N applications will almost certainly lead to a reduction in emissions on average. Crop residues that contain large amounts of N are better left on the surface if possible in order to avoid the oxygen-depleted conditions that arise when the soil becomes

wet. It is also sensible to delay application of mineral fertiliser for at least 4 days following the application of manure or slurry (a rich source of carbon that stimulates microbial activity). Applications to wet soil should be avoided and so should applications when heavy rain is forecast. It is clear that some of these recommendations will conflict with other constraints such as the need to complete a large number of farm operations in a short period of time. Inevitably some kind of management optimisation system will be needed.

The calculation of emissions of nitrous oxide is fraught with difficulties. There seems to be a need to develop more sophisticated means of determining emissions, particularly as a component of the need to contribute to the international effort reporting GHG emissions. Our knowledge of emissions of N_2O from ploughed-up grassland and from vegetable crops is poor and our knowledge of emissions from energy crops is almost completely lacking. We know very little about the leaching of nitrate from under these crops or indeed about their nitrogen cycle in total.

Provision of guidance on the amounts of N fertiliser to be applied to field crops hinges on the interpretation of data from field trials. Despite the large extent of work during the last 30 years or more, this knowledge is still limited in its precision. Guidance is at present based on averages that conceal a very large amount of variation. More sophisticated ways of managing nitrogen application to crops such as canopy management of oilseed rape are suggested as a means to increase capture of N or reduce losses.

One major component of the decision on how much N to apply is the actual and potential supply of N from native soil organic matter. Currently this is approximated by previous cropping, the organic matter content of the soil including application of organic manures, winter rainfall and soil type. In keeping with all of the guidance, this works well on average but conceals a large variation. Even where measurements of soil mineral N are made at application time, post application mineralisation and supply to the crop can blur the precision with which the guidance is judged to work. Several useful attempts have been made to improve our knowledge of soil mineral N supply but for one reason or another, often down to the need for additional information, these have not been adopted widely. A part of the problem seems to be the need to integrate different kinds of information of varying quality. One possible solution to this problem might be by inference using Bayesian Belief Networks (BBN) in which distributed information such as output from a computer simulation model or probability of precipitation can be integrated with discrete information based on ordinal numbers such as field assessment scores of crop vigour or discontinuous information of the yes/no kind. BBNs are often used in diagnostic systems and lend themselves well to increasing precision with each additional piece of information and yet manage to function with relatively little to go on. They are self-learning and can incorporate field and year-specific information along with more general guiding principles such as the mechanisms underlying mineralisation. They integrate the specific with the general.

During the analysis of many field trials assessing the optimum N rate to secure maximum economic yield, it was apparent that yield did not relate to N applied in any simple way from year to year although the economic N rate could be determined in specific trials in particular years. It seems that factors other than N limit yield from year to year as well as from site to site. Post anthesis drought is a source of yield loss in crops but it is apparent that water in the rooting zone can have a significant impact on the yield, development and possibly nitrogen use efficiency (NUE) of an annual crop throughout its life. Even before anthesis and when the soil contains a relatively large amount of water, soil can become too strong for roots to penetrate. Such root restriction may impede the exploration of soil and limit yield later because roots have not developed deeply enough to access water when drought does strike. More than this, root-sourced hormonal signals that derive from impeded roots may restrict or alter shoot growth. Any of these things may affect the efficiency with which N is taken up or used but our knowledge of N use under these conditions is lacking. In winter, roots of young autumn-sown plants may be subject to a shortage of oxygen in wet soils. This is a stress in itself that delays development and may restrict nutrient acquisition at critical times in crop growth. Restricted root growth during winter may mean that crops are more susceptible to drying and to strong soil in the spring. Strong soil appears to be affecting crops now but is not forecast to become worse with climate change. Oxygen stress, on the other hand appears to be a stress on crops that is increasing and may get worse in a substantial part of Eastern England because UK winters are likely to become both warmer and wetter. Taken together, crops growing on half the land area of Eastern England are likely to be affected by one, the other or both of these stresses by 2050. How this might affect N use is unknown. The simplest means to reduce the impact of waterlogging is to apply more N. This, however, is at variance with the need to minimise N_2O emissions.

Precision agriculture promised to revolutionise many aspects of farming a decade ago but failed to deliver in the area of Precision Application (PA) of fertiliser in a spatially-variable way. In part this failure may have been due to a scientific inability to ascribe cause to the effects observed. We know fields are spatially variable and we know that yields are too but it has not proved a simple matter to determine which spatially variable component of the environment determines yield and why. Seasonal variation is usually at least as important a factor as environment and the two interact. Few studies found increases in yield as a result of PA but most suggested that reductions in the total amount of N applied could be achieved without loss of yield. It seems likely that nitrogen use efficiency can be increased by reducing applications where crop growth is expected to be constrained. Increased yield might result from removing the constraints. In order to make progress, however, methodology is needed to acquire specific knowledge on individual fields as to what is causing the variation in yield and infer the appropriate action that is needed.

2. Introduction

The essential remit of the review is at first sight large: to improve understanding of the N cycle in agricultural systems as a basis for improving crop and environmental management. However it is impractical to review such a wide ranging subject in a short period of time and three key areas of interest were identified: (i) strategic nitrous oxide reduction – improved mineral fertiliser N timing strategies, (ii) reducing nitrogen losses from farms – UK potential for nitrification inhibitors and (iii) reduced tillage and N_2O emission.

Since Jarvis et al. (1996) reviewed the mineralisation of nitrogen, there has been much research in N cycling but no general scientific review. Although such a comprehensive review is beyond our scope here, it is likely that there is much more recent UK-based research or research from temperate regions that might be of relevance to UK conditions. We have reviewed the scientific literature relevant to N-cycling focussing on research relevant to greenhouse gas emissions, nitrate leaching and efficiency of use of N.

3. Use of Nitrification Inhibitors to reduce N_2O emissions:

3.1 Background:

Nitrous oxide, N_2O, is a potent Greenhouse Gas (GHG), and its long-term warming potential is about 298 times that of carbon dioxide (Forster et al. 2007). N_2O emissions account for around 10% of global GHG emissions, with around 90% of these derived from agricultural practices (IPCC 2007). For comparison N_2O accounted for 5% of total GHG emissions in the UK with agriculture accounting for 75% of these (McCarthy et al., 2010). Oenema et al. (2005) reported that N_2O emissions from animal waste could be as much as 30-50% of global N_2O emissions.

N_2O is mainly derived from the microbial processes nitrification and denitrification. One of the potential mitigation methods to reduce N_2O emissions from agricultural soils is to use nitrification inhibitors (NIs) to slow down the conversion of NH_4^+ to NO_3^- (nitrification) in the soil, thus slowing down denitrification of NO_3^- to N_2O (Di et al. 2007). Nitrification inhibitors can also reduce nitrate leaching (McKervey et al. 2005).

Many chemicals have been tested as nitrification inhibitors (Table 1), but few are commercially available (Chen 2008). Common NIs include Dicyandiamide (DCD) and 3,4-dimethylpyrazole phosphate (DMPP). Many studies have shown that the application of DCD and DMPP can reduce N_2O emissions from fertilizer (e.g. Asing et al. 2008; Weiske et al. 2001; Zerulla et al. 2001) and organic manures (Asing et al. 2008; Dittert et al. 2001; Hatch et al. 2005).

NIs were originally conceived and evaluated primarily as a way of reducing nitrate leaching and fertilizer N use, rather than as a potential way of reducing N_2O emissions. They have been available as commercial products for several years and are used in mainland Europe, notably Germany, but only to a very limited extent in the UK (NT2601: Dampney et al. 2003). They have generally failed to gain acceptance amongst the UK farming community due to their poor cost-effectiveness in terms of giving yield benefits and reduced NO_3^-

leaching losses (Defra 2007). Furthermore, their effectiveness declines with time, and some NIs may have a toxic effect on some plants (Forbes et al. 2005).

3.2 Recent research developments:

Work in the past on using NIs has focused on reduction of N_2O emissions from N fertilizer and manures. Recent research in New Zealand and Australia has shown that NIs can be extremely effective at reducing N_2O emissions from intensively grazed pasture (de Klein and Eckard 2008). Reviews of the research in New Zealand have been published by groups in Northern Ireland (McKervey et al. 2005, concentrating mainly on nitrate leaching) and for the Pastoral Greenhouse Gas Research Consortium in New Zealand (Suter et al. 2006). In summary, DCD applied to urine patches has been found to reduce N_2O emissions by 61-91%, to reduce NO_3^- leaching by 30-79% and increase annual pasture yield by 0-36% (Clough et al. 2007; Di and Cameron 2002; 2003; 2005; 2008; Di et al. 2007). Rates of application were of the order of 10 kg DCD ha^{-1}. This N-rich compound may thus be expected to supply up to 7 kg N ha^{-1} to a crop itself. This is similar to rates advised for use in arable farming (http://www.omex.co.uk/Agriculture/MoreInfo.asp?id=331&pg=Products). Hatch et al. (2005) found complete inhibition of nitrification in the laboratory but this reduced to 40-60% in the field under Portuguese conditions, in keeping with earlier results from Spain (Macadam et al. 2003).

Kelliher et al. (2007) developed revised emission factors for nitrous oxide emissions from agricultural pasture treated with the nitrification inhibitor DCD in New Zealand. They advise that DCD application to dairy cattle urine should be considered to correspond to a 74% reduction in nitrate leaching ($Frac_{LEACH}$), and a 67% reduction in the direct N_2O emission factor ($EF_{3(PRP)}$). However, the authors acknowledge that these emission factors were derived from a limited number of samples, and that further work is required. They also note that to their knowledge, no other country has revised it emission factors to account for the effects of nitrification inhibitor application to soils.

Nitrification inhibitor coatings on N fertilizers are commercially available and have been shown to be effective in reducing nitrification and N_2O emissions by up to 80% (de Klein and Eckard 2008). Indeed DMPP is only made available in this manner (Table 1). In a novel approach, Ledgard et al. (2008) showed that sheep feed could be supplemented with an NI, with the inhibitor excreted directly in the urine, where it was effective in inhibiting the nitrification of urine-N. Further research is being carried out to examine simple methods of administering NIs, such as a slow-release bolus, and to determine their long-term potential.

In summary NIs show considerable potential for reducing N_2O emissions. Research to date has tested them under 'favourable' conditions such as known urine patches. They need to be evaluated fully under UK conditions.

3.3 Adverse effects

3.3.1 Toxicity.

The current understanding is that there no risk to human health from dicyandiamide or its residues. DCD has been in production for over 30 years and has been widely tested in various countries where no significant detrimental effects have been noticed (Suter et al. 2006).

Dicyandiamide is practically nontoxic to mammals, with an LD50 of 10000 mg/kg (Amberger 1989), and tests have found no mutagenic activity or long-term carcinogenicity .
The acute toxicity of nitrapyrin to humans is low; its aural LD50 is 713 and of the formulation, 1405 mg/kg (AgroSciences 2008) (for comparison, an LD50 exceeding 2000 mg/kg would be considered non-toxic). It is slightly irritating to the eyes and skin and has a low but significant vapour pressure, which is an issue in storage (AgroSciences 2008). In soil and in plants, nitrapyrin is chemically and biologically rapidly degraded into 6-chloropicolinic acid, the only significant chemical residue from its use, and further to N_2, Cl^-, CO_2, and H_2O.
At high doses, however, DCD has toxic effects on plants, or reduces the yield of range of plants. e.g. lettuce (Zerulla et al. 2001), corn, cotton, and sorghum (Reeves and Touchton 1986) and clover (Macadam et al. 2003). It is transported in the xylem and accumulates in the leaves, where it may cause patchy chlorosis (Amberger 1989). In a pot experiment, white clover yield was reduced by DCD application at relatively high rates (25 kg and 50 kg DCD/ha). These are respectively 2.5 and 5 times greater than typical field application rates.
The price of DCD is quoted at £2.95 l-1 (Garry Talbot, Rres Farm, 2009, personal communication), and when added to fertilizer applications at 9l/ha (CC0233: Defra 2001), would add a cost of £26 ha^{-1}.
Nitrapyrin has a selective bacteriostatic and bactericidal effect on Nitrosomonas bacteria, killing part of the soil population (Powell and Prosser 1986). This of course is exactly what nitrification inhibitors are expected to do but it seems unlikely that they will not have an effect on at least part of the wider soil microbial population.
DMPP appears to be superior to DCD and is effective at lower concentrations (McKervey et al. 2005) and is more persistent in soil. Much less has been published on the use of DMPP which is supplied as a compound with the N fertiliser. The ENTEC range of compound fertilizers with added DMPP is marketed in Europe to horticultural and specialist users. ENTEC 26 (26% N, 13% S) is around £330/t (Mr R Willey, Masstock Arable Ltd., 2009, pers. comm.). Because it is only available within compound fertilizers, it is unsuitable for use on urine patches (Kelliher et al. 2007).
3.3.2 Effects on Ammonia volatilization:
Further work is also required to look at the effects of using NIs on subsequent ammonia volatilization. If an NI is applied to a urine patch and it slows down the rate of nitrification and prolongs the presence of ammonia under conditions of elevated pH, it might be expected that such conditions would favour loss of ammonia through volatilization (Kelliher et al. 2007). Although expected, this was not observed in a lysimeter study on a deep sandy soil (Di and Cameron 2004). Chen et al. (2008) note that there is some evidence that the use of NIs may result in increased ammonia volatilization, if surface-applied ammonium-based fertilizers are used.
In summary the NIs discussed here appear to be quite specific in their action on nitrifying organisms. They generally have low mammalian toxicities. LCA data is lacking in general, however

3.4 Natural inhibition of nitrification

Some plants exude compounds that are natural nitrification inhibitors. (Defra AC0206; (Moorby et al. 2007)). Examples include creeping signal grass (*Brachiaria humidicola),* an important pasture grass in tropical regions, and *Leucaena leucocephala* a tropical legume tree. Neem is a natural NI sourced from the Neem tree *Asadirachta melia*. It has been studied and is used mainly in India where positive effects on GHG emissions have been observed (Suter et al. 2006). Incorporation of this type of trait into arable and forage crops grown in the UK would reduce N_2O emissions from applied fertilizer and manures. Genetic modification is one potential route, or conventional breeding, after screening for the trait in native plants that are close enough taxonomically to allow breeding with commercially important crops. Fillery (2007) has reviewed this topic and concludes that current crops do not appear to have the machinery to release inhibitors. Transformation of the plants would therefore appear to be a necessary.

It is possible that the benefits of natural inhibition could be obtained indirectly through undersowing in arable or seed mixtures in pasture. Most of these species might be difficult to manage in farming in the UK because they are (a) tropical species, (b) large, e.g shrubs or trees or (c) invasive.

In summary natural inhibition of nitrification does not appear to be possible in arable agriculture without genetic modification. Pasture mixtures might be a route but there is little work to evaluate.

3.5 Areas for further research and suggestions

CC0272 (Defra 2005) concluded that NIs were a potent, relatively untried means to reduce emissions of N_2O from UK agriculture. A number of questions remain to be answered and issues clarified before wholesale adoption of their use, however.

Many of the studies on NIs reported here have been conducted under relatively ideal conditions for N_2O emissions or leaching losses (Suter et al. 2006) and crucially, for demonstrating the value of NIs. For example, trials were run in lysimeters, receiving high doses of urine and high rainfall inputs, which are likely to represent the ‘worst case’ scenario for losses. NIs were added immediately. On the other hand, annual losses have been estimated from manual chamber measurements, which may have missed peak N_2O emission events (Yamulki et al. 2001) and so emissions may thus be worse than we are aware of. Studies have been conducted at small scale experiments and scaled up to whole field or farm results (Suter et al. 2006), yet scaling is difficult and uncertain. New Zealand grazing paddocks are generally freely draining and have a long growing season. This is very different to the UK situation, where soils used for grassland are predominantly heavy textured and the growing season is much shorter. Situations that would appear to offer most scope for the use of NIs would be soils subject to prolonged wet periods and which are at risk of compaction that exacerbates the retention of water. This would include grazed pastures on clay soils and zero till arable soils receiving mineral N fertiliser. Further work is required under UK field conditions, to determine **if NIs can be as effective as under the conditions reported in New**

Zealand and to see if similar increases in yield and reductions in nitrate leaching as well as N_2O emissions can be achieved

Little research has been carried out in New Zealand comparing the effect of soil type on the effectiveness of Nitrification Inhibitors. Singh et al. (2008) compared the effectiveness of DCD on three soil types and concluded that DCD degradation and thus effectiveness varied with soil properties, such as organic C content and mineralogy. It is also possible that the NI and N substrate may become separated in soil through adsorption, and mass transport. The wider effect of *all* NIs on the functioning of soil organisms is not known. The rate of DCD degradation in soils is strongly dependent on temperature, ie slower degradation in cooler soils. Kelliher et al. (2007) concluded that DCD application is most effective if restricted to months when soil temp was <12^0C. Based on a literature review, they predicted that at soil temperatures of 4, 8 and 12 ^{0}C DCD had a half life of 109, 73 and 49 days, respectively. **A detailed laboratory study is needed to identify the soils and soil properties that favour inhibition, the conservation of inhibitors and full ecosystem impacts of their use. Field work is needed to translate the results into practice and to inform models.**

Most of the work in New Zealand has been carried out with the nitrification inhibitor DCD (Kelliher et al. 2007), and most other NIs have not been assessed for their effectiveness in reducing N_2O emissions from grazed pasture systems. In fertilizer trials, other studies have suggested that DMPP may be a more effective nitrification inhibitor than DCD (Weiske et al. 2001). **Further work is required to assess *other* NIs under UK conditions especially**

Some work has shown the benefit of including the inhibitor within the diet of the animal or as a stomach bolus. Other work has suggested mechanical delivery activated by the lifting of the animal's tail (McKervey et al. 2005). There may also be potential for use in storage of manure and with housed animals. **Novel delivery methods and the use of NIs in intensive areas such as housing and stores need to be investigated**. Note that with this latter ammonia emission may become the critical problem, however.

Prediction of the benefits and persistence of NIs is difficult without a model of their function. Laboratory and field work recommended above should be coordinated and feed into the production of a model of the function of NIs under UK agricultural conditions. If UNFCCC inventory reporting switches to tier 2 or 3 methodology, sufficient data will be needed to develop large-scale models of the effects of nitrification inhibitors and the on-farm means to manage N_2O**. A model of the action and persistence of NIs should be developed and all experimental trials coordinated in a manner that will help with the development and testing of the model.** CC0272 (Defra 2005) recommended that the current emission factors for N_2O emissions should be superseded and validated.

No LCA could be found for NIs and this remains a key lack (Eckard 2007). Some NIs are N rich which presumably means that atmospheric N must be fixed in order to produce them, thus consuming fossil fuel. Others are expensive to produce for other reasons. As seen above they are not without side-effects. LCAs of dairy farming and first generation biofuels that have considered the use of an NI suggest that they may be beneficial. **A full economic**

analysis and LCA of NI use is required in order to evaluate the net benefits of a drive to use NIs in UK agriculture.

Much nitrification and denitrification takes place in the autumn. N rich residues from sugar beet, oilseed rape, ploughed leys and many vegetable crops could be contributing to N_2O emissions at this time. NIs could be applied to residues after harvest. The cooler weeks ahead may preserve the inhibitor for longer than in spring. There is some evidence of NIs reducing crop yields, however. For example, Rahn (2000) reports on a trial with Chinese cabbage in which DCD slightly reduced crop yields presumably because ammonium-N is not sufficiently mobile in soil to meet the large demand during the period of maximum growth. NIs need to be applied at rates high enough to reduce nitrification whilst the crop is small, but low enough not to limit N supply when the crop is growing rapidly (Rahn 2000). **Research is needed to assess the ability of NIs to reduce N_2O emission from crop residues and leys without impacting on the yield of a subsequent crop.**

Table 1: Some of the most common compounds used as Nitrification Inhibitors

Active compound (common / short name)	Brand name	Manufacturer	Ref	Comments	N content
2-chloro-6-trichloromethyl-pyridene (Nitrapyrin)	N-serve	Dow AgroSci. USA	a,b,c,d	High volatility requires special packaging to use with solid fert, can be used with liquid fert	6%
Dicyanamide (DCD) ($C_2H_4N_4$)	Alzon Didin, Ensan	SKW Trostberg AG BASF	a,b,c,d	In production over 30 years, most widely tested and used in Europe. Some phytotoxicity	67%
3,4 dimethylpyrazole-phosphate (DMPP)	Entec	BASF / K+S	a,c,d,e	Sold only as compound fertilizers. Generally more effective than DCD	7%
5-ethoxy-3-trichloromethyl-1,2,4-thiadiazole (Etridiazole)	Dwell, Terrazole	Uniroyal Chem., USA	b,c,d	A fungicide. Relatively volatile	11%
4-amino-1,2,4-triazole (ATC)		Ishihara Chem.	b,c		67%
1-carbamoyl-3-methylpyrazole (CMP)			b,c	Mainly tested and used in Eastern Europe	
N-2,5-dichlorophenyl succinamic acid (DCS)			c		
polyolefin coated urea			c		
Neem	Nimin	Godrej Agrovet, India	a,b	From the Neem tree Azadirachta melia. Produced and studied mainly in India.	

Various Acetylene -producing compounds:			c	Energy-intensive to produce.
calcium carbide			a,c,d	Encapsulation needed for slow-release.
				By-product ethylene may affect growth
calcium cyanamide	STABL-U		a,c	
phenylacetyline			a,c,d	Very expensive
2-ethinylpyridene (2-EP)			a,c,d	Very expensive
Used in Japan, not registered in USA:				
2-Amino-4-chloro-6-methyl-pyrimidine (AM)		Mitsui Toatsu Co.	b,d	
2-Mercapto-benzothiazole (MBT)		Onodo Chem. Ind.	d	
2-Sulfanilamidothiazole (ST)		Mitsui Toatsu Co.	d	
Thiourea (TU)		Nitto Ryuso	d	Carcinogen to mammals
1-amide-2-thiourea (ASU)		Nitto Ryuso	b	

References: a: Suter et al. (2006); b: Trenkel (2007); c: Dampney et al. (2003), quoting Wozniak et al. (1999); d: Chen et al. (2008), quoting Nelson and Huber (1992); e: Zurella et al. (2001)

4. Nitrous Oxide and tillage

4.1 Background

There are serious gaps in our understanding of the formation of N_2O although the EU-funded project NitroEurope (www.nitroeurope.eu) has addressed many of these, especially in natural ecosystems. NEU does not appear to have looked at the effects of reduced tillage on N_2O and there are few reports of work carried out under UK conditions. Losses of N_2O as high as 45.1 kg N ha^{-1} have been found under conducive North American conditions (Rochette et al. 2008). Almost half of the arable land in the UK is under some kind of reduced tillage and this is likely to be weighted towards clay soils. Where clay compaction, reduced tillage, organic matter and poor timing with respect to rainfall all coincide, the potential for emission of nitrous oxide would appear to be large.

4.2 Nitrous oxide and no till / reduced till.

Bhogal et al. (2007); Powlson et al., (2012) found that there is potential for increased N_2O emission following zero and probably reduced tillage, due to increased soil wetness and reduced aeration, although the evidence for this is currently unclear. They concluded that there is a need to establish unequivocally whether reduced/zero tillage practices increase N_2O emissions (compared with conventional tillage). Density of surface soil generally increases with zero tillage suggesting that the reduction in air-filled porosity is probably a cause. Poor drainage will exacerbate this problem, although through drainage of soluble N and other nutrients such as P brings its own problems.

SP0561 (Bhogal et al. 2007) also concluded that although long-term zero/reduced tillage is unlikely to be widespread under UK conditions (due to weed, disease and compaction build-up), rotational ploughing is a common practice. Conant et al. (2007) suggested that as much as 80% of the additional C could be stored in soil as under direct drilling by cultivating every 2 years. But, it is unclear what effect periodic ploughing has on N_2O emissions. Six et al (2004) found that N_2O emissions under zero tillage declined steadily with duration up to 20 years after conversion. They stressed the need to manage nitrogen carefully to achieve these gains. Further work is required to establish whether there is an increase or decrease in N_2O emission under reduced (as opposed to zero) tillage and under a system of rotational ploughing such as is more typical of farming in England and Wales.

Rochette (2008) reported in a review of 25 field studies that compared conventional tillage and no-till, grouped according to soil aeration status estimated using drainage class and precipitation during the growing season. No-till and reduced tillage generally increased N_2O emissions in poorly-aerated soils but was neutral in soils with good and medium aeration. On average, soil N_2O emissions under no-till were 0.06 kg N ha^{-1} lower, 0.12 kg N ha^{-1} higher and 2.00 kg N ha^{-1} higher than under tilled soils with good, medium and poor aeration, respectively. Few results from under reduced as opposed to zero tillage were cited.

Losses of N_2O have been found to be as high as 45.1 kg N ha^{-1} under North American conditions conducive to its production (poorly-drained clay with high organic carbon content under zero tillage, Rochette et al. 2008). In the UK losses of N_2O have been found to be as high as 13.0 kg N ha^{-1} over 77 days (Rochette 2008) but surprisingly Ball et al. (1999) found little effect of deliberate compaction on

N_2O emissions, possibly because the compaction was not extensive enough in the soil. Compaction as a result of reduced tillage and the consequent reduction on air-filled pore space appears to be one of the mechanisms behind increased N_2O emissions with reduced tillage.

Six studies in the UK compared emissions of N_2O from no till and conventionally tilled arable systems (Baggs et al. 2003; Ball et al. 2008; Ball et al. 1999; Skiba and Ball 2002; Smith et al. 2000; Vinten et al. 2002). Four of these studies were on the same site (Ball et al. 2008; Ball et al. 1999; Skiba and Ball 2002; Vinten et al. 2002) in Scotland, on a clay loam soil with 24% clay and impeded drainage. Ball et al. (1999) showed that treatment effects on N_2O fluxes were greater under spring barley than winter barley, with strong tillage effects associated with wet conditions after rainfall, especially under no-tillage. Skiba and Ball (2002) found N_2O emissions were smaller from winter sown crops than from spring sown crops. However, the three rates of N fertilizer application (40, 80 or 120 kg N ha^{-1}) did not produce an expected systematic linear response in either soil available N concentrations or N_2O fluxes. This was attributed to the lack of rainfall in the ten-day period after fertilizer application and therefore very slow incorporation and movement of fertilizer into and through the soil.

Vinten et al. (2002) found N_2O emissions from spring barley were double those from winter wheat during the period from 18 March 1998 to 12 June 1998, 2.44 kg N ha^{-1} compared to 1.27 kg N ha^{-1}. Rates of N fertilizer seemed to have little influence on emissions of N_2O shortly after application. However, late-season emissions in response to rainfall occurred only in plots receiving the highest rates of N. Although N_2O losses from normal ploughing were relatively small (7.37 kg N ha^{-1}), deep ploughing halved emissions (3.7 kg N ha^{-1}), emissions from no-till were a lot higher (23.9 kg N ha^{-1}).

Ball (2008) showed that a cumulative emission of 13.3 kg N_2O-N ha^{-1} over a 12-week period was possible under no-tillage for spring barley, compared to 3.5 kg N_2O-N ha^{-1} from conventional till. Fluxes of N_2O were from 1.5 to 35 times higher from no-tilled than ploughed even where the soil was of similar bulk density.

A study by Baggs (2003) found N_2O emissions two to seven times greater from zero till than from conventional till treatments, when both residues from previous cropping and fertiliser (200kg ha^{-1}) were added. This study was conducted at Wye College on a silt loam soil.
During the first season, 1999 (in which wheat residues were incorporated), the greatest N_2O-N emissions were measured from zero till treatments: 4.0 and 2.5 kg N ha^{-1} from the fertilised and unfertilised treatment respectively, compared with 0.6 and 0.3 kg N ha^{-1} for conventional cultivation. Likewise in 2000 (in which either rye or bean residues were incorporated) N_2O emissions, measured over 65 days following fertiliser application, were greater from fertilised than from unfertilised treatments. The greatest total emissions were measured from the fertilised rye with zero till, with 3.5 kg N_2O-N emitted, compared with 0.6 kg N_2O-N for conventional tillage. There was a significant interaction between tillage and residue: for instance in the fertilised and unfertilised conventional till treatments, emissions from the rye residues were significantly smaller than from beans..

Smith et al. (2000) modelled how the carbon mitigation potential of European agricultural soils was affected if other green house gases (N_2O and NH_4) were included. The potential for GHG mitigation in agricultural soils under reduced tillage was reduced considerably for the UK as a result of an expected increase in N_2O emissions from land under zero and reduced tillage (Bhogal et al., 2007).

In summary reduced tillage probably increases N_2O emissions by increasing the density of soil and reducing the air-filled porosity when the soil is wet. Susceptible soils may see large increases in N_2O emissions under unfavourable conditions

4.3 Improved mineral N fertilizer timing strategies

In theory, tactical applications of mineral fertilizer N that explicitly aim to avoid large amounts of NO_3 in the soil under warm and wet conditions should lead to a reduction in the emission of the greenhouse gas N_2O (Defra 2007). A substantial proportion of the N_2O emissions from productive agriculture is due to the nitrification and denitrification of mineral N fertilizer applications. Such emissions are highly 'event driven' in that high emissions (greater than 0.5 kg N/ha/day) typically occur only during a small number of days when applications coincide with wet and warm conditions in the soil. If such events could be avoided then large reductions in emission would be achieved. Avoidance might be made possible by using soil tests and/or weather forecasts (Defra 2007, p50).

This could be a highly effective strategy to reduce N_2O emissions, as all productive farms in England and Wales could potentially be affected (195 thousand according to DEFRA (2007, p50), and would have the added likely benefit of also reducing NO_3 leaching losses.

Dalal et al. (2003) reviewed the factors controlling N_2O emissions. In arable systems, soil structure (texture), nitrate concentration, soluble and readily available C and water content appear to be the major factors that affect the N_2O/N_2 balance between N_2O diffusion into the atmosphere and its further reduction to N_2 gas. In simple terms the most reliable indicator has been the fraction of the porosity that is occupied by water (see below).

Burton et al. (2008) found that N_2O emissions could be reduced by delaying part of the application of N fertilizer in a warm wet year. Stevens and Laughlin (2002) found that N_2O emissions from grassland soils could be reduced if mineral fertilizer applications were delayed by just 4 days following cattle slurry application. Rozaz et al. (2001) reported that N_2O emissions from irrigated maize were reduced when fertilizer applications were delayed. However, other studies have shown no benefit from splitting the N fertilizer application: Yan et al. (2001) measured no significant effect of split N application on N_2O emissions from maize in a dry year, but suggested that significant benefits would be expected under normal rainfall patterns. Zebarth et al. (2008b) found no benefit of splitting fertilizer N applications to maize in Canada, in crops which showed no yield response to fertilizer N. Soil nitrate concentration only explained a small proportion of the variation in cumulative N_2O emissions, indicating that in this study, other factors were more important.

The rate of N_2O production from nitrification is normally low below 40% water-filled pore space (WFPS), but increases with increasing water content up to 55-65% WFPS (Bateman and Baggs 2005; Dalal et al. 2003; Linn and Doran 1984). Above 60-70% WFPS denitrification dominates (Bateman and Baggs 2005; Dalal et al. 2003) and both N_2O and N_2 emissions occur. N_2 emissions becomes the dominant form of gaseous N loss above 80-90% WFPS. The maximum emission of N_2O occurs at around 60-70% WFPS (Bateman and Baggs 2005; Dalal et al. 2003; Linn and Doran 1984).

Generally the rate of N_2O production increases with increasing NO_3 content in the soil under conditions suitable for denitrification (Dalal et al. 2003; Ruser et al. 2001; Smith et al. 1998). There is evidence that N_2O production increases at a non-linear rate when soil NO_3^- content exceeds crop demand. This has been reported under maize (Chantigny et al. 1998; McSwiney and Robertson 2005) and spring barley (Zebarth et al. 2008a). Under these conditions, and when conditions are also conducive to denitrification, splitting fertilizer N applications could reduce N_2O losses. Data presented by Zebarth et al. (2008a) suggest that lack of crop N uptake accounts for the relative increase in the proportion of mineral N that is emitted as N_2O at high rates of N application. Dalal et al. (2003) state that NO_3^- usually inhibits full denitrification to N_2, increasing the N_2O:N_2 ratio. This lends weight to the view that splitting applications should reduce losses and that correct N fertiliser guidance is imperative.

Nitrification inhibitors have also been found to reduce nitrate concentrations in arable soils after the application of urea and ammoniacal fertilizers, and thus reduce N_2O emissions (see Dalal et al. 2003 for further references). For example, Weiske et al. (2001) reported that the use of the NIs DCD and DMPP in field experiments with barley, maize and wheat reduced N_2O emissions from fertilised plots by 29% and 45% respectively over a three year period. However, the cost-effectiveness of NIs needs further investigation (see above). Furthermore, Weiske et al. (2001) also claimed that the use of these NIs reduced CO_2 fluxes over the three years by 7 and 28% respectively, and increased CH_4 oxidation which, if confirmed in other experiments, could provide further impetus for the use of NIs (Dalal et al. 2003).

In summary, relatively simple management options might reduce N_2O emission substantially but these have not been tested in a systematic fashion. Results and recommendations of any research should be made available to farmers in a leaflet or simple DSS.

4.4 Grassland and animal production systems

Total annual N_2O emissions from cut grassland ranged from 1.7 to 27.6 kg N_2O-N ha^{-1} (Defra 2001). Yamulki and Jarvis (2002) and Pinto at al. (2004) compared emissions of N_2O from ploughed and undisturbed grassland and from compacted (to simulate animal treading) and control soils. Ploughing increased N_2O emission relative to a control substantially under Spanish conditions (20 fold, Pinto et al. 2004) but to a lesser extent in the UK (2.4 fold,Yamulki and Jarvis 2002). Compaction in the UK increased N_2O emissions sharply (3.5 to 4.4 fold,Yamulki and Jarvis 2002)

4.4.1 Nitrous oxide emissions from dung pats

Allen et al. (1996) have estimated that the total annual N_2O emission from dung pats in typical grazing pastures in England ranged from 0.05 to 1.5 kg N_2O N ha^{-1} y^{-1}, assuming a daily deposition per animal of 10 times 2 kg of dung, a stocking rate of 2 animals per ha and 180 grazing days per year. The wide range was related to variations induced by the time of deposition, weather conditions and soil type.

4.4.2 Nitrous oxide emission as a result of treading

Douglas and Crawford (1993) found that N_2O emissions and denitrification rates were up to twice as great in compacted soil than in uncompacted grassland soil. Torbert and Wood (1992) showed that total ^{15}N losses increased by a factor of 3.6 when bulk density of a loamy sand increased from 1.4 to 1.8 t m^{-3} at 60% water- filled pore space.

Oenema (1997) suggests that treading by cattle may easily enhance N_2O emissions from grassland by a factor of two, as indicated by the few available data on the effects of compaction on N_2O in soil discussed before.

Annual N_2O losses from intensively managed grazed grasslands ranged from 10.3 kg N ha^{-1} y^{-1} on the sand to the high figure of 38.5 kg N ha^{-1} y^{-1} on peat soil.

In summary, issues on grassland soil appear similar to arable soils in that wet compacted soils emit the most N_2O. Amounts emitted from the sector may be more, however, because UK grasslands are predominantly found on heavier soils in wetter parts of the country

4.5 N_2O emissions from crop residues

The incorporation of N-rich crop residues into the soil (eg green manures, crop residues) can lead to an increase in N_2O emissions from the soils (Baggs et al. 2000; Möller and Stinner 2009; Velthof et al. 2002) but this source of N_2O is poorly quantified (Velthof et al. 2002). In another study, the application of green manure reduced N_2O emissions, due to temporary immobilization of soil N (Sarkodie-Addo et al. 2003). In a study with 10 different crop residues added to two soil types, Velthof et al. (2002) reported that between 0.1 and 14.6% of the N added as residue was emitted as N_2O; losses were much greater from the sandy soil than the clay soil. The current IPCC default emission factor is 1.25% of crop residue N. Vegetable crop residues may release up to 150 kg N ha^{-1} through mineralization (Chaves et al. 2006).

Gregorich et al. (2005) cite work showing that crop residues left on the surface emitted less N_2O than residues ploughed-in and that legume systems also emitted less N_2O than other arable systems, although the whole cropping system rotation was not considered in all cases. Leaving residues on the surface can promote the transfer of diseases from one season to another. Recent work at Rothamsted (Mahmuti 2009) has shown that diseased crops contribute to GHG emissions by using nitrogen fertiliser less efficiently, thus wasting the fossil fuels during fertiliser manufacture. Thus crop protection may have a benefit to the GHG balance.

NIs have been reported to reduce nitrate leaching from vegetable crop residues, and improve fertilizer N use efficiency (Zerulla et al. 2000 for DMPP), although Chaves et al. (2006) reported that no studies had been carried out to investigate whether NIs could reduce nitrate leaching after incorporation of crop residues. Their incubation study with cauliflower residues reported that under ideal conditions, DCD and DMPP inhibited nitrification from crop residues for 50 and 95 days respectively, with the potential to reduce nitrate leaching (and by extension, N_2O emissions) after incorporation of vegetable crop residues.

Ball et al. (2007) investigated how the ploughing date and cessation of grazing of grass-clover leys in an organic system affected N_2O emissions from the following cereal crop. Losses were 1.2 – 3.9% of the N inputs, equivalent to 5.5 to 8 kg over 17 months.

Residues may reduce N_2O emissions by temporarily immobilising nitrogen in organic forms. N_2O emission from N-rich residues appears to be extremely variable. In some cases it can lead to very large losses. The variability in the published results probably reflects both lack of understanding as well as inherent variability.

In summary N_2O emissions from crop residues appear similar in magnitude to those from fertiliser on a per hectare per year basis, but in general are less well studied. Incorporating later or using NIs may help

4.6 Areas for further research and suggestions

Losses from soil of N_2O often amount to 1 or 2 kg ha^{-1} y^{-1} only and some loss is probably inevitable if land is to be farmed. However, several factors can increase these losses substantially. The most important condition to avoid is wet conditions and compaction in poorly draining soils. Controlled traffic and other measures to alleviate compaction will almost certainly be beneficial (Ball et al., 1999). There are several gaps in our knowledge: most UK studies on the comparison between tilled and reduced tillage soils appear to have been concentrated on one site at Penicuik. **A wider range of soils and climates needs investigation**. Nor have conditions more typical of UK agriculture which **includes rotational ploughing** been investigated. **Splitting applications of N** deserves to be more fully investigated as does the loss from N-rich residues (osr, beet, potatoes, vegetables) **after harvest. There is a lack of information on emissions from under vegetable crops** and emission factors under grassland and arable probably need revisions (Defra 2001). Once rigorously quantified through experimentation, the benefits of reduced tillage need to be fully costed in economic and environmental terms. Current models and emission factors need to be revised in the light of the need to report inventories of GHG emissions. The IPCC emission methodology is widely believed to be inadequate. Very different EFs may apply in autumn from spring.

Defra (2001) suggested the use of urea in spring rather than ammonium nitrate fertiliser and lowering the water table by 30cm in order to reduce emissions of N_2O in grassland systems. The use of urea was more cost effective in this respect but lowering water tables might be effective in reducing anaerobic conditions too, with other benefits such as to the plant roots (see section 6). Climate change and recent increases in the price of fertiliser may change this conclusion. Emission factors for N_2O as a result of the application of mineral fertiliser vary with type of N. Bouwman et al (2002) suggest a slightly higher emission factor for urea than for calcium ammonium nitrate, and the use of urea is also likely to lead to greater emissions of ammonia. NT2601 (Dampney et al., 2003) reports that ammonia emissions from urea application to agricultural land are almost 15 times greater than those from ammonium nitrate. Ammonia emissions may pose environmental health risks and are a source of indirect GHG emissions. When indirect N_2O emissions from ammonia deposition are taken into account the total N_2O flux from urea and ammonium nitrate are likely to be of a similar order of magnitude (Dampney et al., 2003). However, ammonia emissions from urea use and consequential indirect N_2O emissions can be mitigated via the use of urease inhibitors (Dampney et al., 2003). Pathak and Nedwell (2001) compared N_2O emissions from soils at different moisture status treated with different fertiliser types. There is some evidence to suggest that urea is more suited for use on soils prone to water-logging, whilst ammonium nitrate is more suited to less wet soils (Pathak and Nedwell, 2001). Although nitrification inhibitors were effective at field capacity in this study, they were ineffective under waterlogged conditions. Careful choice and use of these fertilisers based on soil type and moisture status is therefore required to reduce ammonia and N_2O emissions. **This underlines the need to include analysis of direct and indirect N_2O emissions (via ammonia losses to air and nitrate leaching to water) in research on the use of urease and nitrification Inhibitors.**

Glendining et al. (2009) found that GHG emissions (including N_2O emission) from farming could be reduced by decreasing fertiliser applications. The benefits to ecosystem service provision was also reduced however, because of the need to farm more land to maintain food production. Glendining et al. (2009) did not specifically examine reduced tillage. Williams et al. (2006) estimated the contribution of N_2O to the total global warming potential of ploughed based agriculture to be 75% and of reduced tillage to be 84%. These are large and imply that action on N_2O will have a large effect on emissions of GHGs from agriculture**. The full economic and environmental benefits of reducing tillage should be assessed.**

Further research is required to determine when it is worth splitting fertilizer N applications to reduce N_2O emissions. A simple farmer friendly indicator tool for avoiding high emission events could be developed, based on measurements of soil NO_3^- content, water content and weather forecasts, to avoid applying fertilizer N applications when soil nitrate content is already sufficient for the crop demand. **The key would be to couple a forecast of rainfall during the coming week with the current state of soil, presenting the information as a risk of loss.** A recent report from the Royal Agricultural Society of England (Godwin, 2008) suggested that field drains were not being maintained and that expertise and extension to the UK industry had been lost. If so, N_2O emissions may be getting worse and much could be gained by restoring drains to full function.

5. Improvements to the provision of guidance on the use of fertilisers to farmers

5.1 Background

Better guidance to farmers on the application of N fertiliser will implicitly save N, reduce pollution and maintain or increase yields at the national scale.

5.2 Potential decline in fertility as a result of the new fertiliser guidance

The UK fertiliser guidance manual (Defra, 2010) suggests that because of the changes to the economics of farming, modern, higher-yielding, more responsive varieties should be given no more N than has previously been advised, despite the fact that they respond to more N. Although this may imply less loss of N, it could lead to a decline in fertility, generally, and fears have been expressed that this might be the case. The full consequences of this change in guidance need to be assessed because we are unable to say with confidence what the effect of reducing fertiliser applications might be. This gap in knowledge could be filled by **a desk study of potential changes in fertility and emissions likely as a result of use of the new guidance,** in order to give confidence in the new guidance and forewarn of potential problems with the revision. Increased soil testing might help with specific field-by-field recommendations but even measurement of SMN is not without problems (see below under estimating SNS

In summary applying less N may lead to a slow decline in soil organic matter. A short desk study could quantify this risk

5.3 Estimating Soil Nitrogen Supply (SNS) to crops

A key component of assessing N application of fertiliser is the supply from the soil. Where possible a measurement of soil mineral N is recommended, otherwise an estimate can be made from the field history. Both methods suffer from the inconsistent nature of the way the data in the databases have been collected and from the need to assess N supply from the soil post-application. There is a need to overhaul and improve the way Soil Nitrogen Supply (SNS) is estimated and interpreted. Such guidance is likely to be of particular value in Nitrate Vulnerable Zones, (NVZs).

Several models of the nitrogen cycle or components of the N cycle, including SUNDIAL (Smith et al. 1996), Well-N (Rahn et al. 1996), NT2501 (Defra 2003a) etc, NGAUGE (Brown et al. 2005) exist. These models were built with the intention of feeding into guidance on the application of N fertiliser to crops and to calculations of the losses of N to the environment. A particularly difficult issue has been estimating the nitrogen supply to crops from soil during the period between fertiliser application and harvest ie after application. SUNDIAL was conceived to provide exactly this information. With the advent of NVZs getting N guidance right has assumed greater importance. A recent HGCA-funded project (Knight et al. 2008) found that soil N supply following fertiliser application could be large and was variable, making refinement of the advice difficult. As a result, nitrogen use efficiency (NUE) was found to vary from about ten to several hundred per cent. Fertiliser recovery can vary from 30 to 72% between vegetable crops. Further measurements and experiments need to be carried out over a balanced range of seasons, soil types and previous cropping. Ongoing HGCA-funded research is acquiring more measurements of soil mineral N but is running up against the problem of field variability. It has highlighted the need for further funding to take measurements of soil mineral N at harvest in order to prove the uptake of soil mineral N, but has not yet demonstrated the unequivocal value of SMN measurements (HGCA 2008).

In general, measurements of soil mineral N before fertiliser application have not been found to correlate as well with optimum yield as is necessary in order to make N application guidance precise (Knight et al. 2008). This appears not simply the issue of the post-application soil N supply, but hangs on what the soil mineral N (SMN) measurement means. In some soils, large SMN can be taken as an indicator of a fertile soil and so much mineralisation to come; in others it implies that the ability of the soil to supply N, derived from the previous year's crop residues has been exhausted. Variability and imprecision of the factors involved mean that process models have sometimes struggled to perform well in practice, although feedback on SUNDIAL and Well-N is almost universally good. Although an empirical model derived from data may explain existing results statistically, it seems unlikely that it would be as good predictively as a simple process model, especially since any database may contain bias. There is merit, however, in an alternative approach that could combined the advantages of empirical models (simplicity, robustness) with that of process models (soundness of concept such as whether high SMN implies much or little mineralisation to come, for example). Such an approach could be met by an Hybrid Bayesian Belief Network (BBN). **There is a significant gap in our ability to integrate the different kinds of knowledge that contribute to the decisions surrounding the application of fertiliser to crops:** historical responses, current soil conditions, recent past cropping, summer mineralisation from soil and so on. Some of these lend themselves well to process models such as mineralisation, some lend themselves better to statistical treatment such as likelihood of good growing conditions, others would seem better suited to measurement or at least inference such as indices or scores such as the Normalised Difference Vegetation Index (NDVI) of N content of the plant at the time of application. Part of the trouble is that the relevant processes operate at different scales: microbial in micropores, macropore leaching and root extraction, establishment and management at field scale. A BBN can incorporate all these different kinds of information; the hybrid term is used where parts of the inference derive from process or other models. Belief networks have found particular success in diagnostics in medicine or fault finding in engineering and computer software. However, they are also capable of quantitative prediction and would seem ideally place to bridge the divide between empirical and process models in order to deliver risk-based guidance on fertiliser use. Risk here we can imply both risk of failing to achieve the expected yield and risk of emissions of pollutants to the environment.

In summary estimating N supply from soil is one of the most complex components of fertiliser guidance. Much effort in this area has improved our understanding greatly but our ability to predict is inadequate. A novel approach to integrating this knowledge in a Bayesian network could be the key to improved predictability

5.4 Canopy management

Canopy management of oilseed rape (OSR) is being suggested by the industry in order to optimise light interception. It is possible to produce too much growth early with a risk of lodging and shading of the lower but productive parts of the crop. Careful use of growth regulators and fungicides with growth regulation properties, and maintaining the correct amount of N in soil for sufficient but not excessive growth, have been found to increase OSR yields by 0.36 t ha^{-1} (Berry and Spink 2009; HGCA 2009). The technology appears to offer significant benefits to farmers in terms of increasing yield by increasing nitrogen use from the same total application but the reasons for smaller canopies have beeb questioned and there is a need to make guidance comprehensive enough to encompass the situations where canopies are already too large or growth was restricted for reasons other than N. The trials to date encompass nine site seasons and it would seem opportune to consider **testing canopy management more widely and more systematically especially in relation to a wide range of canopy sizes in early spring**

In summary canopy management offers the opportunity to increase yields of osr without additional N. Parts of the industry have concerns about the reasons for different canopy sizes and these need to be addressed before the technology can be adopted more widely

5.5 Areas for further research and suggestions

Nitrogen capture from soil and the reasons underlying variability in NUE are two of the biggest unknown components that feed into guidance on fertiliser applications. **A soundly-based, robust means is needed to estimate the soil nitrogen supply** to crops at the time that N fertiliser is applied.

6. The root environment and nitrogen acquisition

6.1 Background

Simple water shortage is not the only or indeed the first consequence of drought. Soils become too strong for roots to penetrate long before crops become short of water (Whalley et al. 2008). As a consequence, roots are unable to explore soil fully and so reach all the water and nutrients present. Root-shoot hormonal signals potentially reduce productivity (Whitmore and Whalley 2009) In contrast, the warmer wetter winters implied by climate change may lead to hypoxia in soils. Crop roots subject to early hypoxia may fail to extend into deep soil early on. This in itself can be a problem for nutrient acquisition and in any case, waterlogged crops usually need to acquire nutrients when the oxygen stress is relieved. Root problems can be compounded if a shallow root system resulting from early hypoxia later tries to extend into soil that is becoming strong. Where crops fail to yield to their potential, it is likely that fertiliser N will not move into grain, posing a potential pollution problem. The record 2008 harvest is probably attributable in part to alleviation of the stresses induced by strong soil that occur in normal, drier years. Understanding these stresses more fully will allow us to avoid over-application of N and also achieve better optimum crop growth.

In summary, direct water and nutrient limitations are not the only impacts of root growth on plant productivity. Strong or wet soil can reduce root growth and lead to yield reductions and failure to access or respond to nutrients. Our winters are expected to get warmer and wetter and our summers drier.

6.2 Areas for further research and suggestions

Freely draining and shallow soils seem to escape the effects of both hypoxia and strong soil but yields are limited on shallow soils anyway

We need to know more about how nitrogen use is impacted by strong soil and which soils are most likely to be affected. Hypoxia seems to have the potential to reduce growth and the use of soil N. Economically the response to waterlogging is likely to be the application of additional N to stimulate tillering. Clearly this is a high risk strategy from the point of view of N_2O emission in soils that are close to saturation. **We need better understanding of the likely impact of hypoxia** and how this will change with climate. A root systems model could be adapted to calculate this more effectively but **experimental data is needed to parameterise models so that they can be used widely for arable crops** in England, Eastern Scotland and parts of Wales. About half of the land-area in Eastern England seems likely to be impacted by strong soil, hypoxia or both by 2050.

7. Spatially variable N application

7.1 Background

Precision application (PA) of nitrogen fertilizer aims to optimise the amount of N applied, thereby increasing margins and decreasing emissions and environmental impact. Goulding et al. (2008) suggest that farmers may be reluctant to adopt precision application technology unless it increases yields. Lark and Wheeler (2003) found savings of £30 ha^{-1} from reduced applications of N but noted that most of the benefit came from reducing applications to unresponsive parts of the field. Robertson (2008) quantified within field variation in the wheat-belt region of Western Australia, and suggested savings of up to \$44 ha^{-1}. Desbourdes et al. (2008) describe Farmstar, a French DSS for fertilizer N and growth regulators, based on satellite images. Over 50% of recommendations at late application (flag leaf) stage to winter wheat were heterogeneous and therefore might benefit from variable-rate spreading. For oilseed rape the proportion was 85%. For the most heterogeneous fields, benefits of up to 0.25t yield OSR, worth £75 ha^{-1}, for no increase in average fertiliser use are claimed. Although relatively recent, all of these studies were carried out under rather different economic conditions from those of today.

Whitmore and Van Noordwijk (1995) pointed out that farmers are acting entirely rationally if they apply a cheap input such as N at the spatially maximum rate to obtain a valuable output such as grain. Logically then reduction of N use should improve NUE and reduce losses, but yields cannot be expected to improve unless significant parts of the field are impacted adversely by high N, perhaps because of susceptibility to lodging. Dailey et al. (2006) suggest that better weather forecasting would help with environmentally-sound fertiliser guidance by helping to avoid yield loss through applying *too little* N**. The potential for a reduction in N_2O emission appears not to have been studied in relation to precision farming**. Simple application of IPCC emission factors (EF) would suggest losses <1 kg ha-1 y-1, but it is likely that over-fertilisation is largely responsible for the currently observed emissions of N_2O and so a reduction in fertiliser might have greater benefits than application of simple EFs suggests. If yields are unlikely to increase with PA and N fertiliser prices remain reasonably low, widespread adoption of PA in order to reduce N pollution will need to be stimulated by other means.

Any one field will have its own degree of spatial variability in response to N that may or may not be amenable to spatial management. Variability in yield (which is what is commonly observed) may be due to measurable properties, some of which are essentially permanent (e.g. soil type & landscape position). Some act over years (e.g. effect of past land use, cultivation and liming), or appear in the current season (e.g. current seed and fertilizer rate). Where properties change, the pattern of N response across the field may be only partially stable from year to year. Even constant properties may cause land to yield differently in contrasting years (e.g. a light, well-drained soil and a heavy clay may be affected differently in wet & dry years). Nevertheless, part of such variability may be resolved into spatial components, which can be summarised into zones within which there is a similarity in behaviour. Differences in the nature of the variability may explain conflicting reports in the literature. Much work to date has focussed on single attributes within a field whereas it is the interaction of several different components of the variation that leads to the observed result. Understanding all field variation is essential to implementing PA.

Thus there is a first general NEED which is the means to determine if, where, how and to what extent spatially variable fertiliser N might benefit either productivity or environmental protection.

So, even where yield is manifestly variable, the causes of this variability are not generally known nor is it straightforward to deduce the optimum management strategy. Indeed information which is available is often available at different scales (temporal and spatial) or resolutions. Thus high-power brute-force computer modelling is unlikely to be helpful or at best will be wasteful.

The second NEED is the means to integrate information from different scales and seasons, ascribe cause and deduce the appropriate intervention simply and in real time, possibly on the go.

7.2 Precision Application in practice

Some of the principles of successful PA for N management were listed by Sylvester-Bradley et al. (1999). They suggested identification of stable zones in fields (persistent variation), and interpreting the properties of those zones in the context of seasonal conditions via an understanding of their interactions, into actions. They suggested soil properties (eg available water, organic matter, pH) are likely to be the most important. However, comprehensive nutrient mapping was thought too expensive at the time, and there was a shortage of suitable sensors to provide information at the necessary spatial resolution.

Clark et al. (2005) briefly reviewed recent advances in measurements and sensors available for *in-situ* and remote measurement of soil and plant nutritional and physical status that would affect plant growth. The more reliable methods for measuring soil moisture and nutrient status require either installation of probes in the field (at limited locations) or taking soil samples (which is expensive).

Much field equipment is now available for precision farming, allowing accuracy in the region of 30cm, and with Real-Time differential Kinematic (RTK) technology, about 2 cm (Cousins and Padfield 2009). Equipment in commercial use includes self-guiding tractors, GPS-controlled rates on fertilizer spreaders and on sprayers, auto–section cut-off spray booms and yield recording on combines. Some of this has been driven by efficiencies due to positional accuracy in field operations and benefits of controlled traffic farming (Chamen 2006). There has been some take-up of precision herbicide use (Lutman and Miller 2007), and various precision fertiliser schemes. As a result technology is in now in place or available for greater use of precision fertilizer application. Current research into the benefits of wider rows to enable mechanical control of weeds as a result of EU restrictions on the use of chemical sprays might facilitate deployment of machinery that delivers N more precisely to plants.

There are various commercially available systems for precision fertilizer application based on crop reflectance; although exact details are not always published for commercial reasons, these use differences between the near infra-red and the visible (often red) light to provide an index that may be related to LAI, crop N or crop dry matter. In general, more N will then be applied to areas of poor growth, and less to areas of strong growth (although the opposite approach may be taken under some circumstances). Satellite image-based systems supply interpreted images to the farm. Usually with some input from the farmer, these are converted into the GPS application maps that control the spreader. Available systems include SkyFarm N-Map and Soyl's SoylSense. Tractor-mounted sensor systems instantaneously determine the application, based on a user-supplied standard fertilizer N rate, adjusted according to the deviation of the local reflectance reading from that of a user-selected reference area.

Simplistic interpretation of the Normalised Difference Vegetation Index (NDVI) could result in local over-application of N (Zillmann et al. 2006). The Yara N-sensor was used in the prescribed manner on a heterogeneous field by applying more N where NDVI is low (to stimulate growth) and less N where NDVI high (to avoid lodging). They found this approach led to poor FUE and surplus soil mineral N in areas coinciding with shallow soils. This reveals a limitation of reliance on the sensor approaches alone when yield-limiting factors. Scotford and Miller (2004) concluded that once the canopy has closed, further growth of LAI will have little effect on the spectral reflectance of the canopy, and additional measurements (e.g. height) are needed to quantify crop growth. They propose ultrasonic technology to do so.

In summary, there is much technology available to support PA but it has tended to be driven by what is possible. The issue is deciding what spatial information is useful to acquire, interpreting it and inferring actions. It is unlikely to be possible to acquire knowledge of all the underlying variation in a single growing season.

7.3 Role of historic data

A modelling exercise on zone management of nitrogen, (Brennan et al. (2007) suggested that prior knowledge of weather was more important than knowledge of spatial differences in field although knowledge of both contributed to economic performance. Furthermore they point out that economic optima may be aligned even though yield responses differ. Lark and Wheeler (Lark and Wheeler 2003a; b) and Lark et al. (2003) measured the response of a winter wheat crop to N over entire fields, during two different growing seasons and determined the economic optimum. They found significant differences in the yield response but the practical value was generally in reducing applications to unresponsive areas. This would save 30 kg N ha^{-1} or about 40 euro ha-1 at 2003 prices. It is possible that there are reasons for the observed lack of response to N. If these could be identified and ameliorated then profitability might increase. **In general, results imply that PA will reduce over-application. That it reduces losses of N has not yet shown to be the case.**

Lark et al. (2003) tested the hypothesis that regions of a field, defined by the fact that their relative yields vary between seasons in the same kind of way (which may include consistently good or consistently bad), are subject to similar limitations. In all fields, some soil properties were found to differ significantly between the regions so-identified. This was put forward as the basis for a method to identify spatially-responsive parts of fields and to apply PA accordingly. Lark et al. (2003) admit that not all fields may contain sufficient spatial variation to make PA worthwhile. Their methodology, however, is able to detect and eliminate non-responsive fields. Their large survey (34 fields all of which were spatially responsive) lends greater weight to the idea that PA may help than the smaller surveys of Welsh et al. (2003a; b) and James and Godwin (2003) who surveyed one, two or three fields respectively. Welsh et al. (2003a; b) found that increasing fertiliser to parts of the field that had low NDVI (to estimate shoot density, in early March, immediately prior to N application) derived from satellite data gave the best increase in yield. The value of such a survey then is to establish the variation in economic response to fertiliser N in order to develop a strategy for PA.

In summary, historical data can be used to firstly to exclude spatially consistent zones and secondly to indentify zones where spatially variable applications might help. In most cases this will lead to a reduction in the amount of N applied and by implication less loss of N

7.4 Factors limiting yield other than N

There may be factors other than N supply that limit yield. If so, the simplistic approach of applying more N where growth is poor may be ineffective in increasing yield and risk environmental losses of N. Clearly any factor important to plant growth or its response to N that varies at the field scale could be of relevance in PA. However, it may be difficult to discern such factors, or quantify their effect. In general NUE will be improved by removing other constraints to crop growth. Removing such constraints may lead to an increase in yield and the focus on N alone rather than N use may explain why PA has not been found to increase productivity to date. Clearly a review of all such constraints and their alleviation is beyond the scope of this report but a brief summary of the issues and techniques to assess the spatial variability of such constraints seems appropriate.

Vrindts et al. (2005) were able to define, for a field, zones where soil bulk density (ie compaction) limited yield. Outside these areas other factors (eg soil moisture variability) had more effect on yield. Water supply, pH, pest or disease pressure or inadequate supplies of other minerals could all put restrictions on the ability of a crop to respond to N. Making the plant monitor itself is possible by the use of reporter genes. For instance the sdi1 gene has potential for reporting sulphur deficiency. Potentially a transgenic plant could be used to reveal the nitrogen status of the soil as experienced by the crop (Howarth et al., 2009; http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=10553#Description). Soil water potential is of great importance to crop growth; it can be estimated from soil physical properties and meteorological data, but is difficult to measure remotely (Clark et al., 2005). The importance of soil strength to crop performance is discussed elsewhere in this review. Here it is noted that plough draft force (Watts et al. 2006) and a profile of penetrometer force (Sudduth et al. 2008) can be measured spatially. Cultivator-mounted reflectance spectroscopy shows potential for measuring various soil properties including soil organic matter and pH (Christy 2008), but may require calibration for each soil type. Yield may be limited by areas of disease in the crop and these might be alleviated economically by precision spraying. Oliver et al. (2003) showed that hyperspectral imagery could potentially be used to detect patches of take-all in wheat fields. Take-all patches will have a lower response to N. Variation in take-all may be reasonably persistent if the fungus can only survive in crop residues that remain where they grew. Likewise (Chen et al. (2007) found that, for wheat fields in China, NDVI derived from Landsat could be used to assess disease prevalence.

Schellberg et al. (2008) reviewed precision agriculture of grassland, and concluded that this discipline is not yet mature. In particular, it is not clear what and how to monitor, for instance soil properties, nor is it clear how to interpret the measurements made and deduce an action. Schellberg et al. (2008) also suggest that appropriate models are needed.

There may be uses for PA technology other than for application of fertiliser N. Examples include the frugal use of herbicides on weed patches; nematicides, where over-application is costly and environmentally damaging; fungicides, where the rate could be linked to the canopy density in the field (denser canopies are both more prone to disease, and have more surface to cover in product).

In summary, crop yield may respond to spatially variable factors other than N. Alleviating these constraints in a spatially variable way might increase yield while spatial application of N may reduce environmental losses. Equipment might find uses in alleviating these constraints as well as in applying N.

7.5 Areas for further research and suggestions

There is much disagreement in the literature and in practice about whether PA can benefit arable agriculture and if so, by how much. It seems likely that the reasons for this disagreement lie partly in the fact that some fields are spatially variable and will respond to PA while some are not. Not all studies have asked the right question: we need knowledge of how the economic optimum, NUE and N loss vary spatially. This is not the same as studying yield variation or even trying to remove constraints. It seems sensible to characterise zones in a field that will behave differently, however, and to treat them accordingly, bearing in mind that the zones or their responses may change from year to year. To derive benefits, amenable fields and zones within fields must first be identified and their response characterised.

The extent to which losses of N can be reduced with PA is unclear. It does seem likely, however, that N_2O emission and NO_3 leaching can be reduced, since several studies have found no loss of yield with a reduction in total N applied. Savings in these terms have rarely been identified but could be substantial if most emissions derive from the under-use of applied N (perhaps 1-2 kg N_2O-N ha^{-1} or 50 kg NO_3-N ha^{-1} y^{-1}). To date such savings have not been measured in the field.

In addition, developments elsewhere may permit or stimulate the uptake of PA technology. For example, restrictions on the use of pest and disease controlling agents by the EU have led scientists to consider planting crops at wider spacing in order to manage weeds and disease spread mechanically. The deployment of robotic or precision instrumentation would facilitate very precisely controlled fertiliser application.

7.4.1 Deployment of PA in practice

Lark and Wheeler (2003a,b) show how spatially variable N response can be mapped experimentally. (Lark et al. 2003) show (i) how the scope for variable management can be assessed and (ii) how potential management zones (ie regions with similar limitations) can be defined. In addition they deduce proxy measures for acquiring expensive spatial information as economically as possible. If implemented on particular fields during several years, a body of knowledge will build up that would enable a land manager to infer the spatially-variable, optimum economic response to N. To be of use, however, such information must be available at the time fertiliser is to be applied. **To this end Lark and Wheeler's methodology needs to be tested in field during several years in conjunction with the most promising canopy measurements**, probably Scotford and Miller's (2003) measure of shoot number and NDVI acquired remotely.

The issue of limits to yield other than N is beyond the scope of this review. However, the farming community may be reluctant to take expensive measures that reduce N loss unless they also increase yield or reduce other costs too. Implementing the methodology proposed above offers the opportunity to acquire diverse information about the behaviour of the crop and field. Application of field-scale of decision support has not proved successful to date. What is needed is a system tailored to the particular constraints in each region of the field and which is based on mechanistic understanding of how those regions interact with season dependent factors such as weather and previous management. Thus conceived, the system should (i) identify where N limits yield, (ii) where other factors are responsible and what the factors are and (iii) recommend specific, spatially and seasonally dependent interventions that optimise the economic amount of N to be applied while minimising the emission to the environment. Such information would have to be acquired during several years of in-field experimentation.

8. Energy Crops

8.1 Background

We know relatively little about the ways energy crops grown for biomass, such as willow, miscanthus and poplar respond to N. Nor do we know how soil fertility under such crops will change with time. In particular, we do not know what will happen to the large reserves of organic N present under grasslands in the west of the UK if these are turned over to biomass crops that do not take-up the large amounts of N the soils contain. The long-term effect of these crops on soils is completely unknown.

Christian et al. (2006) found similar recoveries of labelled N applied to Miscanthus as applied to nearby cereals. Clifton Brown et al. (2007) estimate total carbon mitigation during 15 years experiment to be 5-7 t C ha^{-1} y^{-1}. N supply was met from the soil but the saving in C does not include any additional N_2O emission from the land-use change. Goodlass et al. (2007) found increased levels of nitrate leaching during the establishment and removal phases of short-rotation coppice (SRC) (60-70 mg NO_3-N l^{-1}, with a peak of 134 mg l^{-1}) compared with autumn leaching under arable crops (50 mg l^{-1}) but reduced leaching under the established coppice itself each autumn (18 mg l^{-l}). Jorgensen et al. (1997) concluded that N_2O emissions from fertilised Miscanthus or rye were a minor problem at about 1.5% emitted from under the Miscanthus of that applied and 0.5% from under the rye. The Miscanthus soil had been under an orchard 4 years previously and the rye was grown on an arable field. Kavdir et al. (2008) found smaller proportions of total N_2O emissions from applied N (33% of total) compared with arable crops (67%).total emissions were greater from the arable that SRC (4.3 kg ha^{-1} versus 1.9 kg ha^{-1}) also. They concluded that N_2O emissions derived mainly from nitrification rather than denitrification. Pinto et al. (2004) found emissions of 0.56 kg N_2O-N ha^{-1} during 5 days following the ploughing of perennial grassland. Bertora et al. (2007) found 1 mg N_2O-N kg^{-1} soil emitted during 62 days following disturbance of grassland. This is approximately 1-2 kg ha^{-1} depending on density and depth. Whitmore et al. (1992) report losses of up 4 t ha^{-1} N on ploughing permanent pasture during two or three decades. Much of this was assumed to leach but IPCC EFs would suggest 50 kg N_2O ha^{-1} in total weighted heavily towards the first few years.

8.2 Areas for further research and suggestions

Much N is released when permanent pasture is ploughed up. Energy crops are likely to replace grassland in the landscape before they replace other crops. We do not have good data on the movement of carbon or nitrogen below-ground under short rotation coppice or Miscanthus. Grassland production is more prevalent on wetter heavier soils in the West of the UK; research needs to be focussed on the conversion to energy crops of soils in these areas. We need to know if levels of N under energy crops are similar to those under grass and crucially whether it **is possible to maintain organic carbon and nitrogen levels in former grassland soils during the transition to energy crops in the wetter western parts of the country**. Any loss of CO_2 or N_2O from soil will reduce the GHG balance and thus usefulness of these crops. Findings should be incorporated into models for generalising to the whole of England and Wales.

Energy crops were included in the Defra fertiliser guide for the first time in the recent revision (Defra 2010). It was apparent during the revision that there is insufficient information on the response of energy crops to fertiliser and to nitrogen fertiliser in particular. **Energy crops should be included among those which are examined for their response to applications of N and a better**

understanding of the nutrition and cycling of N under these crops sought through experimentation and modelling.

In summary grassland soils currently contain much organic C and N. Even if soil under energy crops can store large amounts of organic matter (and we do not know this yet) there is no guarantee that reserves in former pastures will be preserved on conversion to biomass crops.

9. N-Rich Organic Wastes

9.1 Background

Application of certain N-rich organic wastes to land has been reported to increase N_2O emissions (Möller and Stinner 2009) N_2O emissions depend on many factors in soil (under investigation in NitroEurope) but the incorporation of N-rich crop residues, manures and industrial wastes would benefit from general guidance on timing and manipulation and understanding to underpin that guidance.

Möller and Stinner (2009) conclude that digestion of crop residues primarily from biogas production followed by application of the residue to land significantly reduced N_2O emissions. Stevens and Laughlin (2002) recommend adding slurry to soil before adding mineral N as a means to reduce N_2O emissions. Zhang et al. (2008) recommend using low C and N containing soils as covers of landfill as a means to reduce N_2O emissions. SP0530 (Defra 2009) is due to report soon on the addition of farm wastes to soil and their effect on soil quality. One important finding is that much of the benefit from sustained addition year on year derives from the active organic matter in the current year of application.

In summary organic wastes contain N and this N may be at risk of conversion to N_2O

10. Subsoil N

Bradley et al. (2005) report the distribution of soil Carbon in England and Wales. They give data from which the average density of carbon in the subsoils (30-70cm) can be deduced as 0.5% C by weight. There is no distribution or coefficient of variance reported, however and so it is difficult to interpret this further. A typical C:N ratio for soil is 10 and so this implies very crudely that the average density of N in the subsoil is 0.05%, or half the threshold that Sylvester-Bradley et al. (2001) suggest might contribute significant amounts of N through mineralisation to the soil nitrogen supply to crops. Without a reported distribution or other measure of the variation it is unwise to speculate widely on the prevalence of soils containing more than 0.1% N but it is unlikely to be greater than 25% of all soils. On the other hand a soil containing 0.05% N in the soil volume from 30 to 100cm may supply up to a third of the N contained in the topsoil and so about 20 or 30 kg ha^{-1} yr^{-1}. Salomé (2009) have published data that suggest that subsoil organic matter (OM) is as reactive as topsoil organic matter but that it does not decompose unless the soil matrix holding it is disturbed. These inputs have been taken into account implicitly in the current methodology to estimate fertiliser N applications. The data underlying this methodology and subsequent guidance is rather variable, however. A fuller analysis of the N supplying power of subsoils could reduce this variability. The mean amount of mineral N measured in soils at the time of application that supported the recent revision of the fertiliser guidance was about 70 kg N ha^{-1}. Many soils contained much more than this. It is likely, however, that it is the large contributions of N from subsoil, ie where much organic N is present, that will help explain most variation.

In summary there is no evidence that the organic N in subsoils differs from that in topsoils. It may, however be protected by the matrix. Such N could be contributing to crop N supply but we know little about subsoil N reserves and supply.

10.1 Areas for further research and suggestions

Salome (2009) have published data that suggest that subsoil organic matter (OM) is as reactive as topsoil organic matter but that it does not decompose unless the soil matrix holding it is disturbed. **We do not know what mechanisms preserve this OM nor what actions might release it.** With such process knowledge we will need a fuller survey of subsoil C & N in order to predict nitrogen supply and the potential for losses

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