WP 1 Deliverable 1.2

Report on literature findings related to the effectiveness of techniques to reduce N losses to water from agricultural systems

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Date: 15 September 2010
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Acronyms

BAT - best available technique
BMP - best management practice (synonymous with BAT)
CAFO - centralized animal feeding operation
ILF - intensive livestock farming
ILO - intensive livestock operation (synonymous with CAFO)

$N_{\text{min}}$ - nitrogen in $\text{NH}_4$ and $\text{NO}_3$ forms rather than in organic compounds – also often known as available N as these forms are immediately plant available.

N – nitrogen
$\text{NH}_4^+$ - ammonium
$\text{NO}_3^-$ - nitrate
1. Introduction

Nitrate leaching is a complex function of land use, cropping system, soil type, climate, topography, hydrology, animal density and nutrient management. (Kirchmann et al., 2002)

The movement of nitrates into groundwater and surface water from agricultural sources has been identified as a major environmental and health issue within the European Union. The Nitrates Directive (91/676/EEC) was adopted in December 1991 in order to address this issue. The Water Framework Directive (2000/60/EC) was more recently implemented and one of its key aims is to achieve “good status” for all waters by 2015. The Directives particularly focus on preventing the eutrophication of fresh and marine waters (and associated risks to human health), which has become a major problem in many regions of intensive agricultural production in Europe.

The challenge of reducing losses of nitrogen (N) to water due to human activities will not be met unless we fully understand the processes controlling the cycling of N, specifically reactive N (Nr), within the global environment. Reactive N is nitrogen in the form of ammonia (NH₃), nitrous oxide (N₂O), nitric oxide (NO), nitrate (NO₃⁻) and nitrogen oxides (NOₓ) which, when released into the environment, trigger a series (or cascade) of effects that can negatively impact on human and ecosystem health (2008). The proportion of reactive N within the global N cycle dramatically increased after the early twentieth century when the Haber-Bosch process was invented. This allowed a massive increase in the industrial production of Nr globally, which was used to produce crops for a growing world population (Sutton et al., 2009). But the global nitrogen cycle has consequently now become one of the most “perturbed” global nutrient cycles. In comparison to the carbon cycle, which is said to be perturbed less than 10%, the nitrogen cycle is estimated to be 80% perturbed (European Science Foundation Standing Committee for Life Earth and Environmental Sciences, 2008). This perturbation has resulted in an excess of reactive nitrogen (Nr) in the environment.

The role of agriculture

Nitrate losses to water (the sum of losses due to leaching and runoff) are controlled by a range of factors. The soil type, texture, structure and landscape can impact on the amount of nitrate lost. Weather patterns, particularly rainfall volume and pattern, can also determine the extent of nitrate losses to water. Losses of nitrate due to these natural factors can be exacerbated by manmade interventions, such as field drainage, which can increase N leaching in otherwise slowly draining soils by increasing both rates of nitrification and drainage amounts (Stockdale, 1999). Additionally, the amounts of N cycling in most agricultural systems are much greater than under natural ecosystems, so the potential for loss is correspondingly higher (Batey, 1999).

Farm management practices have a major impact on losses of N to water. For example:

1) Synchrony of N availability and crop N demand is critical to restrict losses of N by leaching or runoff. The efficiency of N capture by crops can be affected by fertilization and tillage practices, as well as crop rotation and variety selection. During periods of high crop demand, NH₄⁺ and NO₃⁻ (mineral N) levels in the soil are low and hence, even if drainage
occurs, losses will be low. However, fertilization in excess of crop demand or mineralization of N from crop residues or manures can result in large amounts of mineral N accumulating in soil. Nitrate is particularly susceptible to loss from soil during periods of excess rainfall (i.e. where rainfall is greater than evapotranspiration).

2) Intensity of livestock farming also impacts on N losses to water. High stocking rates on grazed land can lead to more N being supplied in manures than the grass can take up, so that the surplus N is lost by leaching. Producing livestock on farms with a limited land base and a reliance on imported feeds, can also lead to production of manure N in excess of the farm’s N needs, and an increased risk of N losses to water (and atmosphere) unless manures are handled carefully.

Figure 1 illustrates the pathways of N loss to water from agricultural production systems. Strategies to reduce N losses to water from agriculture are usually targeted at the components of the system indicated in the boxes on the diagram or at processes which link them indicated by the arrows.

**Figure 1. Simplified diagram of main pathways for N leaching and runoff losses from agricultural systems (adapted from (2009)).** Black arrows indicate movement within the farm system, red arrows indicate movement of N to water.
The N-TOOLBOX project was conceived as a response to the challenges associated with implementing the Nitrates Directive at the farm level in Europe. Its overall aim is

*to develop a “toolbox” of cost-effective technologies to be implemented at the farm level to protect water from nitrate pollution.*

This report contains the outputs of Task 2 of N-TOOLBOX WP1. The objective of this task is: **To summarize the findings of the literature relating to technologies to reduce N losses to water from agricultural systems.**

2. Survey methodology

We began by accessing the literature published in the previous five years (covering 2004 to the beginning of 2010) through searches on the database ISI Web of Knowledge (Thomas Reuters, 2008-2010, London, UK). The searches were structured by combining the standard words ‘nitrogen’, ‘N’, ‘nitrate’, ‘leaching’, ‘los*’, ‘eutrophication’, ‘nonpoint source pollution’, and ‘diffuse water pollution’ with words covering specific production systems, aspects of these production systems, possible techniques and/or specific regions, in particular following up key practices or systems identified in D1 (Report on findings from EU and national (UK, Denmark, Spain, The Netherlands) projects relating to nitrate contamination of groundwater and surface water from agricultural systems). As the study was intended for application in systems in Mediterranean or temperate European climates, the focus was on research undertaken in Europe, although reviews and research articles reporting work done in North America, New Zealand or Australia were also included if relevant. Conference proceedings and unpublished results were not included in this review.

In addition, the reference lists of relevant articles were searched and cited reference searches of relevant articles performed using ISI Web of Knowledge. This created a snowball style search and it is anticipated that most of the key papers from the past 30 years will have been accessed. The references together with their abstracts (and pdf versions of the papers where accessible) are stored in an Endnote library for use by the project team. We regard the sample of articles found as representative and non-biased and consider that it covers the best techniques available to reduce N losses to water from agricultural systems. However, due to ‘publication bias’ that results in non-significant studies being published less often than those reporting significant results (Arnqvist & Wooster, 1995), it may be that the effectiveness of the techniques is exaggerated and consequently the review may indicate that techniques work better than they generally do.

The following report should be considered as a critical commentary and synthesis of the literature contained in the Endnote library. The report is structured according to the flow chart of possible mitigation strategies outlined in D1 which was developed from the pathways diagram (Figure 1). Each section reports on a mitigation technique or closely grouped set of management approaches and after an introduction provides information on the reported effectiveness, the observed limitations or factors affecting implementation, and identifies any further research needed before the technique can be implemented in practice.
3. Effectiveness of techniques for mitigation of nitrate losses to water from farms

3.1 Dietary management

Introduction

In livestock systems, the proportion of N consumed that is used directly to support production of meat or milk (that is the N use efficiency) is relatively low:

- Beef systems have N use efficiencies of 10% or less (Rotz, 2004)
- N use efficiencies in lactating dairy cows vary between 15 and 30% (Aarts et al. 1992, quoted by van der Stelt et al., 2008; Castillo et al., 2000).
- Taken overall intensive dairy farming has low N use efficiencies, with ruminants excreting 75-95% of their N intake (Whitehead 1995 and Søegaard et al. 2001; cited by Eriksen et al., 2004).
- In intensive pig production where the protein needs of the animals can be more closely met, N use efficiency may average 30 to 35% and even approach 40% (Jongbloed et al., 1997).

Theoretical maximum N use efficiencies vary with animal species, age, stage of lactation, etc. but this theoretical limit is about 50% (Rotz, 2004). Consequently, even in the best managed systems much of the N present in the feed will be excreted by the animal.

A range of approaches have been taken to reduce the excretion of nutrients (N and P) in manure. Nitrogen excretion is directly related to the animal’s N (protein) intake, so to increase efficiency less protein must be fed per unit of production. A common goal in feeding all animal species is to provide the right amount and quality of protein to maximize production at a minimum feed cost. Although improved productivity can increase N use efficiency, greater improvements are generally obtained through strategies that improve protein-feeding efficiency. The content and the digestibility of the feed affects both the N composition and N content of the excrements (van der Stelt et al., 2008). N consumed in excess of animal requirement is excreted in faeces and urine (Kirchgessner et al. 1994; cited by Hoekstra et al., 2007). Urinary N excretion is affected more by dietary change than faecal N excretion (Kebreab et al. 2002 as quoted by van der Stelt et al., 2008; Borsting et al., 2003) and this balance can impact the risk of N losses (Paul et al. 1998, quoted by van der Stelt et al., 2008).

For ruminants it is important to synchronise the supply of N and energy to obtain optimal rumen fermentation and minimal N excretion (Borsting et al., 2003). Amino acids taken up by microbes in the rumen can be incorporated into microbial protein if the supplies of readily available energy (mainly water-soluble carbohydrates) are sufficiently high (Hoekstra et al., 2007). If the supply of readily available energy is relatively low, either amino acids or structural carbohydrates in the diet are used by rumen microbes for the bulk of their energy supply. As these compounds degrade relatively slowly, there can be a lack of both balance and synchrony in N and energy release in the rumen. This leads to the accumulation of ammonia which is adsorbed across the rumen wall and subsequently converted into urea (Nocek and Russell 1988; Miller et al. 2001; both cited by Hoekstra et al., 2007). This urea is mainly excreted through the urine which may lead to excess N in the environment (Hoekstra et al., 2007). The partitioning of N loss over urine and faeces is important to consider as N in urine is a much more important source of pollution than faecal N (Van Horn et al., 1996, cited by Hoekstra et al., 2007).
To date, strategies have largely been developed to optimize feed utilization and reduce excretion of nutrients in ILOs; however, managing diets for efficient use of nutrients also makes sense for more extensive management systems. Four strategies are common for housed animals:

1. Phase feeding where different diets are fed during different stages of the animals’ growth or production.

2. Formulating a balanced feed with an optimum feed conversion ratio based on digestible phosphorus and amino acids (following the ideal protein concept).

3. In ruminants, approaches also seek to optimize rumen function by reducing the degradation of protein in the rumen, enhancing rumen protein synthesis and maximizing endogenous urea recycling.

4. Improving feed characteristics, for example through the use of specific amino acids and related compounds; use of feed additives; sensible application of growth promoting substances; increased use of highly digestible raw materials.

Although it is possible to reduce N losses to the environment through changing the composition of livestock diets, this is more complex for grazing livestock than for animals kept indoors (Hoekstra et al., 2007). Low efficiency of protein utilization is one of the main challenges associated with diets consisting of grazed grass. The main cause of this is thought to be impaired rumen function due to relatively high concentrations of soluble protein and imbalance in the carbohydrate and protein supply (Beever and Reynolds, 1994; Lantinga and Groot, 1996; both cited by Hoekstra et al., 2007). Changes in grassland management can be used as part of dietary management strategies to balance and synchronize carbohydrate and N supply in the rumen and increase the proportion of rumen undegradable protein (RUP).

**Effectiveness**

Diet modification by formulating diets closer to requirements has been shown to reduce N excretion by 10-15% in chickens and pigs (Nahm, 2002). Cole et al. (2006) showed that feeding beef cattle a diet with 11.5% crude protein (around 88% of the typical levels used) provided adequate metabolizable protein to maintain yield but reduced total N excreted and reduced the estimated N volatilization losses by approximately 22%. Le et al. (2009) also showed that reducing dietary crude protein from 15 to 12% of the diet for finishing pigs very significantly reduced total N and ammonium concentration in excreta. However simple reductions in dietary crude protein don’t always reduce N excretion; the impact of changes in diet depends on interaction with energy contents and digestibility of energy sources (Clark et al., 2005). Managing protein contents during grazing is more difficult than for housed animals as the the protein contents in herbage vary depending on environmental variables and herbage management (Hoekstra et al., 2007) and there is currently no rapid on-farm tool for the measurement of protein content in herbage.
Balancing and synchronizing carbohydrate and protein supply to the rumen implies the optimization of both the amounts of rumen available carbohydrates and protein and their respective degradation rates in the rumen (Hoekstra et al., 2007). In ruminants, there are indications that reducing the digestibility of the diets, reduces the amount of N excreted. For example, N utilization efficiency increased when lactating cows were fed grass silage diets enriched with a low-degradable instead of a high-degradable starch source (Castillo et al 2001, quoted by van der Stelt et al., 2008). In forages this would mean increasing the proportion of N related to the cell wall as opposed to soluble N and especially non-protein N (Beever and Reynolds 1994; cited by Hoekstra et al., 2007). However, as pasture often has relatively high protein contents, it can be difficult to find supplements that can balance the N content of the diet (Kuipers and Mandersloot 1999, quoted by Borsting et al., 2003). Restricted grazing, where cows are kept indoors for longer, increases the opportunity to provide low digestibility supplementary feeds.

Research has confirmed that increasing the proportion of rumen undegradable protein in the diet of dairy cows reduced rumen ammonia levels and significantly decreased urinary N excretion with no or only a small decrease in milk yield (Dinn et al 1998 and Castillo et al. 2001b, cited by Hoekstra et al., 2007). This approach is only effective where total protein levels in the diet are not in excess of animal requirements. However, increasing the proportion of cell wall protein in forages (which is less degradable) may increase the proportion of undegradable protein in the intestines, leading to an increase in faecal N excretion (Buxton 1996; Valk et al. 1996; cited by Hoekstra et al., 2007). Although this would not increase the bovine N efficiency, it may still reduce N losses as faecal N is less prone to volatilization and leaching than urinary N (Van Horn et al 1996; cited by Hoekstra et al., 2007).

Van der Stelt et al. (2008) showed the interacting impacts of energy and protein levels in the rations of non-lactating cows on both the amount and form of N excreted. Increasing dietary crude protein levels increased total available and organic N levels in slurry; increased dietary energy also increased N excreted but reduced N lost by volatilization (van der Stelt et al., 2008). In contrast a study with lactating dairy cows showed that increasing the dietary energy content decreased the total N excreted (Broderick 2003, quoted by van der Stelt et al., 2008). Increasing the water-soluble carbohydrate content of forages has been proposed as an important way to improve N utilization of forages since their N content is generally high as well as being highly soluble and therefore rapidly degradable (Hoekstra et al., 2007). Hence dietary modification to reduce N excretion also needs to consider maintenance requirements and interactions with animal age; dietary N efficiency has also been shown to increase with dairy cow age (Borsting et al., 2003). In work on commercial dairy farms, Arriaga et al. (2009) found that manipulation of dietary crude protein levels to better match requirements of cattle was possible and taken together with other management strategies to increase herd productivity, the changes made were estimated to reduce herd N excretion by 11% on a per hectare basis.

Changing the diet of animals during their different growth or reproductive stages is known as phase feeding. It is relatively common to group animals according to level of production and/or growth stage and to adjust the diets for these groups. Phase feeding has been shown to reduce N excretion.
by 10% in chicken and pigs without loss of production (Nahm, 2002). However, Cole et al. (2006) showed that if phase feeding (directed at reducing crude protein intake) for beef cattle is not carefully implemented it can have adverse effects on animal performance as well as leading to significant reductions in N excreted. However, modest changes in dietary contents of crude protein in the later finishing period may have relatively small effects on the overall performance of beef cattle (Cole et al., 2006). Automated and computerised feeding systems, which can feed varying amounts of two or more feeds, can be used to adapt the feeding to the production level of individual animals (Borsting et al., 2003).

Dietary supplements using synthetic amino acids have been shown to reduce N excretion by 10-27% in broiler chickens, 18-35% in laying hens and 19-62% in pigs with the reductions dependent on the actual % reduction in overall protein inputs as a result of the targeted use of the specific amino acids. Other growth promoting substances have been shown to improve N use efficiency significantly but only by small amounts (Nahm, 2002).

There are a number of grassland management tools which affect C and N dynamics and therefore also the composition of the herbage and hence grazing livestock diets (Hoekstra et al., 2007):

1. Increasing the length of the re-growth period of herbage before defoliation, reduces the amount of dietary N and increases the ruminal N and energy balance. This leads to a reduction in losses of ammonia-N from the rumen and subsequently also decreased urinary N excretion (Hoekstra et al., 2007). The effect that the regrowth period has on RUP is unclear. However, it is possible that the reduced organic matter digestibility may decrease energy availability with high maturity, which may lead to reductions in animal productivity (Hoekstra et al., 2007).

2. Reducing the N fertilization of fresh forage is an efficient means of lowering urinary N excretion of cows with little or no modification in their performance. Decreased N fertilization leads to reductions in grass crude protein content and solubility as well as an increase in the water soluble carbohydrate content (Nowakowski 1962; Reid and Strachan 1974; Wilman et al 1976b; Smith 1973; Wilman and Wright 1978; Mc Grath 1992; Valk et al 1996; all cited by Hoekstra et al., 2007). This is likely to result in a better balance and synchronization of N and C supply in the rumen. A reduction in N fertilizer application rate will also increase the proportion of rumen undegradable (by-pass) protein (Van Vuuren et al 1991; Valk et al 1996 cited by Hoekstra et al., 2007).

3. In theory, grass varieties with increased amounts of sugar are a promising means of improving N utilization by grazing livestock. However, this has not been seen consistently in animal trials so far, although urinary N excretion tends to be lowered. The total N intake when eating high sugar grasses may be increased through an increased intake rate, because of higher palatability (Jones and Roberts 1991, cited by Hoekstra et al., 2007), lower resistance to physical breakdown during chewing (Groot et al., 2006) and higher digestibility. The increase in water soluble carbohydrate content is likely to improve the balance and synchronization of the N and C supply to the rumen. No effect on the N fractionation (i.e. proportions of cell wall and cell content proteins are similar for several cultivars) or the proportion of RUP has been reported.
Factors affecting implementation

Implementation of any dietary management strategies often remains a challenge as profit margins for livestock enterprises are relatively low and the direct economic return for saved nutrients is small when assessed by fertilizer replacement value. Consequently the major constraint is usually economic, especially where capital investments are required (Rotz, 2004). Available labour is also often heavily used in animal production, so changes that require more time, particularly for the farm manager, will not be readily accepted (Rotz, 2004).

There is a large variation in the amino acid composition between commonly used feedstuffs (Borsting et al., 2003 data from Hvelplund et al. 2001). This requires good estimates of the supply of microbial amino acids, the amount of digestible undegraded amino acids in different feedstuffs and the requirements for amino acids by individual animals or groups of animals (Borsting et al., 2003). Increased requirements for testing of feed ingredients and/or increased numbers of diet formulations to be calculated for each unit together with lack of knowledge of the impacts of changes at farm scale may also be a constraint to adoption of new practices.

Manipulating grassland management to meet targeted dietary aims is complex. Management approaches used on grassland interact and may also affect herbage yields and animal production. Further complexities lie in the fact that effects and interactions are site specific and may vary from year to year. Hence the lack of certainty in the impact of any management practices on dietary composition is likely to mean that this approach is not used except where it is associated with certain benefits for livestock productivity by increasing herbage yield or improving forage quality.

Further research needed

There are no rapid methods to obtain measures of rumen degradability and intestinal digestibility for individual amino acids (Borsting et al., 2003). These are required for some feeds, to correctly estimate the absorbed amount of individual amino acids.

More research is needed to identify the ideal profile of absorbed amino acids for a range of livestock types at different stages in their lifecycle and at different production intensities. In this way, ideal protein supplements can be composed which can minimize N excretion (Boisen et al. 2000, quoted by Borsting et al., 2003).

Further research is required on the effects of herbage management on the fractionation of N compounds in herbage (cell wall or cell content) as well as the effect of high sugar grass varieties on bovine N efficiency under a range of herbage management combinations (Hoekstra et al., 2007).

There is currently little evidence that directly relates changes in N excretion to actual environmental impacts. Work looking at the impacts of dietary manipulation for housed dairy cattle on greenhouse gas emissions suggests that management of slurry on application may be of more importance than slurry composition per se in reducing losses (Arriaga et al., 2010).

Due to the complexities of managing grazing systems, modelling is vital to quantify and predict the effect of any combination of herbage management tools under specific circumstances (Hoekstra et al. 2007).
Unfortunately, it is not yet possible to predict animal N efficiency under grazing from current modelling studies. A model on a grazing system would have to include both herbage and animal components. Combining these separate models is very difficult due to software incompatibility and differences in input and output factors.

Infra-red thermography has shown promise as a method to screen cattle for feed use efficiency. These findings might have application in selection programs and in the better understanding of the biological basis associated with productive performance (Montanholi et al., 2010).

### 3.2 Manure storage and handling solutions

**Introduction**

The definition of manure, for the purposes of the N-TOOLBOX project, includes any organic material that supplies organic matter to soils. This can include sewage sludge and composts made from household waste, and green waste, as well as livestock wastes. Within the N-TOOLBOX project we will use the terminology associated with manure management as defined in the RAMIRAN Glossary of Terms on Livestock Manure Management (Pain & Menzi, 2003).

It is essential that manure storage and handling systems in ILOs are optimized to minimize N losses; however, all farms that have livestock may risk contaminating water supplies from manure. For this reason, strategies that are suitable for ILOs may also be useful in more extensive systems where livestock are housed for some of the year, and in any systems where manures are brought-in or stored and handled on farm.

Nutrient contents and forms in manure are dependent not only on the livestock and their diet (as described in Section 3.1) but also on the bedding materials and also on any additional feedstocks used during processing or composting. Because of the focus of this review on losses of N to water, the extensive literature concerning losses of ammonia gas (NH₃) by volatilization from manures in housing and storage is not covered in detail here — however, strategies which seek to prevent losses on NH₃ (e.g. acidification, housing design; (Rotz, 2004) will tend to lead to manures with higher mineral N contents during storage and at application than those where such strategies are not employed.

Many of the BMPs relating to the improving nutrient use efficiency from manures applied to land require more timely and more accurate applications (Section 3.5). The main strategy currently implemented by EU countries to deal with “the manure problem” has been to implement strict restrictions on dates when spreading is allowable. This requires that livestock farms have sufficient and appropriate storage for manure during the periods when spreading is prohibited. Better practices during manure storage have also been the focus of work carried out to reduce NH₃ volatilization losses (Monteny & Erisman, 1998; Sommer et al., 2006). Leaching losses during storage are a relatively small proportion of the total N loss (usually not more than 10%) during this period. However, changes in amounts and forms of N during storage can have significant impacts on N losses when the manure is applied to land.
Manure treatment is still relatively rare; Oenema et al. (2007) found that less than 6% of manure was treated in any way during storage. However, a number of approaches have been used and are increasingly recommended including:

1. **manure separation** - This is a physical separation of the liquid and solid components of manures to produce different fractions which can then be handled, stored and used separately (Hjorth *et al.*, 2010). After separation, the solid fraction can potentially be used as a soil improver, organic fertilizer or as a component of a compost-based plant-growth medium, potentially replacing peat-based products (Jorgensen & Jensen, 2009). The liquid fraction (supernatant) produced after mechanical separation followed by a sediment settling treatment contains low concentrations of solids and nutrients (Fangueiro *et al.*, 2008). This provides benefits in terms of handling, odours, necessary storage volume, partial or total waste treatment, land application and fertigation (Burton and Turner 2003; cited by Fangueiro *et al.*, 2008).

2. **composting** - Solid manure from confined animals is collected and actively turned to increase aeration and promote controlled decomposition (often for phyto-sanitary reasons). About 40% of dry matter is lost during the composting process which reduces subsequent haulage requirements. High temperatures achieved during the process are able to eliminate most viable weed seeds, coliform bacteria and human parasites (Larney *et al.*, 2006). Up to half of the manure N is lost (dominantly by NH$_3$ volatilization) during the process and the remaining N is stabilized. The implications of these changes in N forms in compost during application to agricultural land and impacts on N leaching will be considered in Section 3.5.

3. **biogas production** - Methane (CH$_4$) is produced for energy through anaerobic digestion of livestock wastes, often with other feedstock materials. The digester effluent/anaerobic digestate is often applied to land. This is a controlled and containerized process with no implications for leaching during the process. The use of anaerobic digestate as a manure applied to land will be considered in Section 3.5.

**Effectiveness**

Leaching of effluents from manure storage systems is negligible when the ground is sealed and the drainage water collected. However, a number of storage methods are used on farm depending on investment of the farm in manure storage facilities, the dry matter content of the manure and amounts of manure that need to be stored. Solid manures are often stacked in a heap. Oenema *et al.* (2007) reviewed all the available information and considered that N leaching losses from solid manure in unsealed heaps are expected to be around 2% of N in the manure when covered, and around 5% when the heap is uncovered. Covering the stacked manure can also reduce NH$_3$ volatilization losses especially where NH$_3$ absorbent materials such as peat are used; covering the heap with straw had little impact on overall N losses (Rodhe & Karlsson, 2002). Storage of slurry takes place in pits or specially constructed tanks/lagoons. N leaching losses from slurries in unsealed lagoons are around 5% of the N in slurry when covered, and around 10% when uncovered (Oenema *et al.*, 2007).
Separation of different fractions from manure with knowledge of their physical and chemical characteristics, can increase the options for manure management (Fangueiro et al., 2008). Hence application strategies can be adjusted to crop need more precisely (Sorensen and Thomsen 2005 cited by Jorgensen & Jensen, 2009; Section 3.5). The studies on manure separation that were found during this literature review focus on identifying the chemical and biochemical characteristics as well as the organic N degradation kinetics of the products from different separation technologies and manure types (e.g. Jorgensen & Jensen, 2009; Fangueiro et al., 2008) and they did not determine the amounts of N lost during the application of this strategy.

Similarly there have been few studies comparing the risk of leaching arising during the composting process. Parkinson et al. (2004) showed an increase in the NO$_3^-$ content in manures as the compost matured. In the study of Larney et al. (2006), composted manure from a beef feedlot had a lower NH$_4^+$ content and higher NO$_3^-$ content compared with the other treatments (fresh or stockpiled), which may lead to increased leaching risks if windrows are uncovered during maturation. Early leaching losses from composting windrows were dominated by NH$_4^+$; small but increasing losses of NO$_3^-$ in the leachate were observed later in the composting cycle (Parkinson et al., 2004); overall only around 1% of the total N added to the windrow was lost in leachate; total N losses were around 30% of the initial N in the heap but mostly occurred in gaseous forms.

Factors affecting implementation

The key issue affecting the implementation of more effective manure storage is the capital investment needed on farm to improve storage facilities. The recent review of the implementation of the Nitrates Directive (Commission of the European Communities, 2007) highlighted manure storage capacity requirements as one impediment to the effective implementation of the Directive. Regulations and incentives related to storage conditions: e.g. covered storage areas, concrete storage pads, have been implemented in a number of national action plans.

As for implementation of dietary management changes, the key issues affecting implementation are labour requirements and requirements for non-profit generating capital investment at farm scale.

Further research needed

Prohibited periods for manure spreading are not well defined by research evidence. Currently storage is required for periods from 2 to 12 months, with large variations even in neighbouring regions with similar climatic conditions. Studies have suggested that minimum capacity should be defined according the required length of storage periods (i.e. prohibited manure spreading periods) ranging from 4 months in Mediterranean areas to 9-11 months in boreal areas. Cost benefit analyses which integrate the environmental implications of different storage and /or manure treatment options is needed to help inform the decisions of both farmers and policy makers.
3.3 Management for reduced N losses on grazed land

Introduction

Cut grassland systems usually have low nitrate leaching due to their long growing season and high N efficiency (Simmelsgaard 1998; cited by Eriksen et al., 2004). Nitrogen losses by leaching (as well as gaseous losses of N) from grasslands dramatically increase where grazing animals are present (Jarvis 2000 cited by Eriksen et al., 2004). Both the duration of grazing and the grazing intensity (stocking rate) will affect the amounts and timing of nitrogen losses to water. For example, in the experimental study of Eriksen et al. (2004), cropping sequences where grazing occurred for longer periods had significantly higher leaching losses than those where grazing was shorter or where cutting took place. During grazing, most of the N excreted is deposited in the field in localized patches at rates up to 1000 kg N ha\(^{-1}\) for cattle (Haynes and Williams 1993; cited by Eriksen et al., 2004). Consequently the amounts of plant available N in those locations exceeds plant demand very significantly and the spatial mismatch between N supply and demand increases N leaching risk. Urinary N has been shown to make up 70-90% of the N leaching from grazed grassland (Monaghan et al., 2009). In addition, mineralization of soil organic matter from grassland soils releases nitrogen. For example, nitrate from soil organic matter was often measured to be in excess of 300 kg N ha\(^{-1}\) yr\(^{-1}\) in the study of Gill et al. (1995; cited by Jarvis, 2000). Actual leaching rates also depend on interactions with weather so that analysis of long-term data has found that leaching from grassland soils is higher in the autumn following dry summers (Scholefield et al. 1993, cited by Tyson et al., 1997; Jarvis 2000) where drought reduces plant growth and hence N uptake so that larger amounts of soil nitrate accumulate in soil and are available for loss.

Trackways are a key part of grazing systems, particularly in dairy systems where cows are brought from grazing to the milking parlour regularly. Geotextile and gravel pads have been proposed as alternatives to concrete to minimize mud, runoff and erosion in heavy traffic areas (Singh et al., 2008).

The presence of streams or other waterways within or adjacent to the grazing area also increases the risks of direct N losses to water, as well as losses via nitrate or dissolved organic N in runoff and leaching. Exclusion of livestock from streams by fencing prevents direct excretion into or near the waters but often also requires the provision of in-field watering points for the livestock.

Sward composition i.e. the inclusion of legumes as a protein source, can also affect nitrate leaching from grazed land. Nitrate leaching is lower from unfertilized grass-clover swards than from mineral-N fertilized swards grazed by either sheep (Ruz-Jerez et al. 1995; Cuttle et al. 1998; cited by Eriksen et al. 2004) or cattle (Owens et al. 1994; Tyson et al. 1997; Hooda et al. 1998 cited by Eriksen et al. 2004). Eriksen et al. (2004) compared N leaching between unfertilized grass-clover (perennial ryegrass (Lolium perenne L.)/white clover (Trifolium repens L.)) and perennial ryegrass fertilized with 300 kg N ha\(^{-1}\) yr\(^{-1}\). Over a period of five years, the average nitrate leaching in the grass systems was 47 kg N ha\(^{-1}\) compared with 24 kg N ha\(^{-1}\) in the grass-clover systems (i.e. a reduction of 49% in leaching losses; Eriksen et al. 2004). Although the herbage production was less in grass-clover
compared with grass, nitrate leaching per unit herbage was still lower in grass-clover than in grass. In a similar study by Tyson et al. (1997), in which the grass pasture received 200 kg N ha\(^{-1}\), average N leaching over an eight year period in the grass and grass-clover pastures was 50.2 and 12.9 kg ha\(^{-1}\) yr\(^{-1}\) respectively (i.e. a reduction of 74%). However, stocking rates were greater in the grass pasture (Tyson et al. 1997). It is thought that there is a natural feedback mechanism that regulates the N\(_2\) fixation by pasture legumes (Eriksen et al. 2004). This mechanism is thought to be driven by soil inorganic N levels. Under low soil N, legumes dominate and derive most N from N\(_2\) fixation. Under high soil N, grasses have a competitive advantage over legumes and dominate (Ledgard 2001; Spatz & Benz 2001; cited by Eriksen et al. 2004), reducing the amount of N\(_2\) fixation. Thus, this feedback mechanism limits the N inputs from legumes and consequently regulates the risk of N losses (Eriksen et al. 2004). However, when swards are compared by assessing leaching losses in terms of the total N input or on a per unit product (milk, meat) basis, it has been shown that there is no difference between amounts of N lost by leaching (Jarvis 1992; cited by Tyson et al. 1997).

**Effectiveness**

Reducing the length of grazing periods (either daily or seasonally) may be an option to reduce NO\(_3^-\) leaching. This would need to be accompanied by provision of separate housing/loafing areas where manures can be collected (as described in Section 3.2). Interactions between the materials from which loafing areas or trackways are constructed and the strategies to clear accumulating manures determine the types and amounts of N lost. For example, Singh et al. (2008) showed that mud plots generated leachate with the highest NO\(_3^-\)-N levels during manure removal experiments (between 0.1 and 0.5 mg m\(^{-2}\)) while the plots with geotextile and dense gravel aggregate generated leachate with the highest NO\(_3^-\)-N levels during manure accumulation experiments (between 0.3 and 2.7 mg m\(^{-2}\)).

Changes in daily patterns of grazing may also affect pasture intake and hence animal production. Clark et al. (2010) compared the impacts of daily grazing patterns in a dairy system on pasture intake, milk production, body weight and body condition score and on location of urinations. The proportion of urinations on pasture and trackways was reduced from on average 87% in the free grazing control groups to 53% in the groups with reduced access to pasture (either one 8 hour period or two 4 hour periods) (Clark et al., 2010). Direct impacts on N losses were not measured.

On an annual basis, the balance between housing and grazing has a significant impact on the form of N losses from livestock systems; so for example, losses due to volatilization are reduced substantially when cattle are grazing relative to when livestock are housed (Webb et al., 2005). Using data from a farmlet study, de Klein and Ledgard (2001) showed that in nil grazing systems (where forage was cut and carried to housed livestock) nitrate leaching losses may be reduced by 55-65%; however, total losses of N were 10-35% higher than under conventional grazing because of increased gaseous losses. In restricted grazing systems (where grazing does not take place when net drainage is occurring) leaching losses of N were reduced by 35-50% with no overall change to the total losses of N from the system. In a desk study combining various models, Webb et al. (2005) studied the impacts of extending the length of the cattle grazing season in the UK on ammonia and nitrous oxide emissions. The findings suggested that if the grazing season were extended by another month, assuming that the cattle would graze for an average of 4 h per day in this extended period, the
annual reduction in NH$_3$ emissions may only be ca. 1-2% compared to the current grazing season (around 180 days). For most of the modelled scenarios, all N conserved from loss via volatilization was predicted to be lost via NO$_3^-$ leaching instead. In the case of FYM and where slurry was applied to clay soil, the increases in NO$_3^-$ leaching were always greater than the NH$_3$ conserved. The effects on direct emissions of N$_2$O were estimated to be negligible. It is likely however, that there would be an increase in indirect N$_2$O emissions due to the increased NO$_3^-$ losses (Webb et al., 2005).

Nunez et al. (2010) measured the impact of factorial combinations of grazing intensity and grazing frequency on N losses during grazing compared with a no grazing treatment. While the frequent heavy grazing strategy increased all losses – increasing grazing frequency had a larger impact on leaching, while increasing grazing intensity had a larger impact on NH$_3$ volatilization. Rotational grazing strategies have been found to restrict nitrate leaching, as they distribute grazing over a relatively large area so that hotspots with repeated or overlapping urinations are rare (Haynes & Williams, 1993). Where rotational grazing is combined with the restriction of the grazing period to times when net drainage is not occurring then very significant reductions in N leaching losses can be achieved without loss of production in intensive dairy systems (Boumans et al., 2001). Rotation of cutting and grazing across the whole grassland area of the farm can also reduce grazing frequency; however, this may not be practical as different sward types are often preferred for grazing and conservation. However, losses of N from grazing fields still contribute a large proportion of the total N lost by leaching from a dairy system even where best grazing management is in place (Verloop et al., 2006).

Vidon et al. (2008) studied the effects of cattle grazing on water quality along a stream section where near stream vegetation was lacking and stream bank erosion due to cattle activity was obvious. However, stream monitoring did not show any increased nitrate load directly due to cattle access. However, increases in ammonium and total N concentrations were especially high when the flow was low and cattle were more often observed in or near the stream (Vidon et al., 2008). Restriction of livestock access to surface waters also reduces sediment and phosphorus inputs to the stream (Cherry et al., 2008). Miller et al. (2010) found that providing in-field watering points and preventing access to the stream increased soil contents of nitrate at depth compared with both watering points immediately adjacent to the stream and control areas. This highlights both that watering / feeding points act as focal points within a field for the congregation of livestock and tend to create hotspots of N (as well as phosphorus and potassium) as a result of excretion but also that ensuring that such points have relatively low connectivity to the water course can reduce the rate (if not the total amount) of N lost to water.

Factors affecting implementation

Because of the interactions between grassland productivity and farm profitability, and the interaction between housing and grazing for livestock, strategies to reduce N leaching from grassland are complex to study and also to implement.
Access to pasture has been shown to improve cow health and welfare (Clark et al., 2010), hence reducing grazing periods could potentially have adverse effects on cow welfare, depending on the specific circumstances.

Reducing the length of the grazing day will not reduce (and may increase) labour requirements without any increase in production (Verloop et al., 2006). Any changes to grazing management systems which lead to increased labour requirements or a need for capital investment, such as provision of in-field watering or modification to trackways, are unlikely to be adopted due to economic constraints without a corresponding an increase in production.

Reducing the length of the annual grazing season would require higher amounts of supplementary feeding either as conserved forages or concentrates. For dairy cows this might increase costs by £8 and £40 cow\(^{-1}\) year\(^{-1}\) (Webb et al., 2005). In addition, there would be an increase in the volume of manure produced inside or around farm buildings which would need to be appropriately stored and spread.

**Further research needed**

There are no simple monitoring tools available for farmers which indicate the likely onset of net drainage and its relationship to late season grass production. Without such knowledge farmers will not be able to determine most appropriate time to cease grazing as the dates of drainage onset are season-specific; this is most critical on soils susceptible to leaching such as sandy and shallow soils (Verloop et al., 2006).

### 3.4 Predicting optimum N application rates for crop and pasture land

**Introduction**

The determination of crop and pasture N requirements in the growing season is usually based on yield potential expressed as a farm/field-specific yield goal. A range of information can be used to guide each farmer’s yield goal, including regional averages and previous yields achieved at the site. This yield goal is used to determine the likely crop N demand. To obtain a recommended N application rate, it is also necessary to predict the quantity of N that will become available during the growing season from crop residues, previously applied organic fertilizers and soil organic matter (often these sources are together termed the soil N supply). However, the actual achievement of both the yield goal and the predicted soil N supply will be affected in each growing season by weather, crop disease and a range of other factors (Goulding et al., 2000).

N response curves describe the relationship between crop yield and increments of added N fertilizer. The response usually shows a typical pattern with diminishing returns for each subsequent addition of N and eventually the achievement of a plateau (or even a yield reduction). Economic optimum rates are determined as those where the price obtainable for the yield increase achieved is greater than the cost of the additional N supplied. In a study where different rates of fertilizer were applied to corn, Stevens & Quinton (2009) observed that high fertilizer N rates beyond an optimum led to decreased uptake of soil-derived mineral N resulting in an increased accumulation of unutilized....
mineral N in the soil at crop harvest. For cereal crops significant additional N leaching does not seem to occur until after the economic optimum fertiliser rate (Goulding et al., 2000). However, at rates above the optimum N application, leaching loads increase substantially (Bergstrom and Brink 1986; Lord and Mitchell 1998; cited by Kirchmann et al., 2002). In practice, rates used by farmers may be well above optimum; for example, in a study by Scholfield et al. (1993), the amount of nitrate leaching loss over a seven year period from permanent grazed swards receiving 200 and 400 kg fertilizer N ha$^{-1}$ yr$^{-1}$ was on average 38.5 and 133.8 kg N ha$^{-1}$ yr$^{-1}$ respectively. The increase in N input from 200 to 400 kg ha$^{-1}$ led to only a marginal, uneconomical, increase in animal output (Tyson et al., 1993; cited by Tyson et al., 1997). Due to the uncertainty involved in predicting the economic optimum, it has been suggested that any N application rate should be decreased to below the expected economic optimum rate to reduce the risk of large N leaching losses (Kirchmann et al., 2002).

Relatively little work has been carried out to support farmers in determining appropriate yield goals. In most N recommendation systems, the requirement to predict the likely weather, disease risk and other factors potentially impacting on crop growth is explicitly placed with the farmer or their advisor. However, some work has been carried out to allow the adjustment of yield goals in season and hence to affect the rates of N applied in split applications through the use of plant based diagnostic tests which give an indication of both crop N demand and current N supply simultaneously (Zebarth et al., 2009). These include:

1. Determination of the total N concentration for key plant parts and comparison with known critical values for different plant organs and growth stages (Olfs et al., 2005); Greenwood et al. 1990, reviewed by Zebarth et al., 2009).
2. The sap nitrate test which is a field-based and semi-quantitative means of determining crop N status (Olfs et al., 2005).
3. Measurement of chlorophyll content of a plant is a good qualitative indicator for its N concentration (Olfs et al., 2005). This is usually determined using a correlated measure of “greenness” commonly using a SPAD-502 chlorophyll meter (Minolta Corp., Japan). Ideally, the use of reference plots (either well-fertilized or not fertilized) should be used together with plant tests so that an in-season N management strategy can be applied (Samborski et al., 2009).

A range of approaches have been used to predict soil N supply and hence to adjust N application rates including:

1. N credit systems which give an indication of N supply from preceding crops, manure applications and soil where values from tables are used to allow for soil N supply (Zebarth et al., 2009).
2. Measurement of soil mineral N (Nmin). The concept behind Nmin sampling is that crop response to N fertilization will generally be lower in soils with high soil mineral N and vice versa. Recommendations of when samples should be collected, which layers to sample and how to adjust N-fertilizer rate, differ substantially between countries (Olfs et al., 2005).
3. Estimation of mineralizable N through chemical analysis or biological incubation (Sharifi et al., 2007). A range of approaches have been tested but none are in use routinely for N recommendation.
4. Integration of information using computer-based decision support systems. A range of computer-based recommendation systems have been developed to aid in determining N fertilization rates for specific farms. These systems integrate large amounts of experimental data and scientific understanding and include both static (database look-up) and dynamic simulation models e.g. (Dampney et al., 2000; Veihe et al., 2006; Bleken et al., 2009; Cichota & Snow, 2009).

Effectiveness

Research on the use of the range of approaches to predict N supply tends to focus on reducing the quantity of N-fertilizer application whilst maintaining similar or increasing crop yields. For example:

- Bundy and Andraski (2004) tested a range of diagnostic tests and demonstrated that where nitrogen rate recommendations for wheat were adjusted for pre-plant soil NO₃⁻ contents excessive N applications were avoided at 11 of 21 sites, thereby increasing economic gains due to reduced fertilizer costs and avoiding yield reductions due to excessive N availability.
- Zhao et al. (2007) showed that the use of SPAD readings to adjust fertilizer N rates reduced wheat yield spatial variability and lead to a low soil residual NO₃⁻N content.
- Dailey et al. (2006) showed the small change in crop N uptake but significant economic benefit for arable farming in the UK that could be obtained by the use of more accurate medium term weather forecasting in the determination of crop N demand.

The reduction in N application might be used as an indication of the reduction in N leaching, but there are few data where N losses to water have been measured or predicted alongside the use of different recommendation approaches.

Simulations have shown that where the Nₘᵢₙ test is used to measure pre-planting mineral N in mixed rotations including vegetables and using this information to adjust the standard N recommendation, leaching losses could potentially be reduced by 66% compared to typical farming practice which uses fertilizer rates corresponding to the crop’s respective target value minus 20% (Nendel, 2009). The simulation using the EU-Rotate_N decision support tool took account of various different vegetable-producing model farms in the Baden-Württemberg region, Germany. If all the farmers in the region were to apply typical practice, N losses would correspond to 202 kg N ha⁻¹ yr⁻¹. If all farmers in the region were to apply the crop’s respective target value minus the amount of soil mineral N available before sowing or planting, N losses would correspond to 69 kg N ha⁻¹ yr⁻¹.

Davies (2000) states that although restriction of fertilizer use appears to be the most obvious measure to reduce N leaching, it is unattractive upon further examination. Farming profit comes from the last few per cent of the yield response to fertilizer N. According to Davies (2000), if N application is restricted to less than the economic optimum, it is this part of the yield that is sacrificed. In addition, even large restrictions of fertilizer N use would initially have comparatively small effects on reducing N leaching. Only after many years will N loss reductions be substantial, as organic N reserves decline (Davies & Sylvester-Bradley, 1995).
Factors affecting implementation

The major disadvantage of all site specific sampling approaches is that they require additional costs and labour (Nendel, 2009).

Field sampling of plant tissue and determination of total N concentration with sufficient detail to guide fertilizer practice is expensive and time-consuming and is therefore not currently considered appropriate as a practical tool (Olfs et al., 2005; Zebarth et al., 2009).

The use of N$_{\text{min}}$ (either as a pre-plant or pre-sidedress test) is only economically advantageous compared with using fixed N rates if a considerable variation in soil mineral N exists between fields and from year-to-year (Neeteson 1989; Everett and Pierce 1996; Schröder et al. 2000; all cited by Olfs et al., 2005). In the study by Nendel (2009), costs for N$_{\text{min}}$ sampling and analysis were assumed to be € 25 ha$^{-1}$ sample$^{-1}$. The mean annual gross margin for all rotations was calculated as € 6375 ha$^{-1}$ yr$^{-1}$ and € 6269 ha$^{-1}$ yr$^{-1}$ for typical fertilizer practice and the use of N$_{\text{min}}$ test respectively. In general, the costs of using the N$_{\text{min}}$ test only affected the gross margin marginally.

Further research needed

Lemaire et al. (2008) highlighted the need to develop remote sensing measurements which can allow indirect estimation of crop nitrogen status. Tractor-based sensors could be used at the same time as other field operations are carried out e.g. Yara N-sensor.

Rapid assessment methods for soil indices are also being developed (e.g Murphy et al., 2009) and these should continue to be evaluated under field conditions.

The current N fertilizer recommendations take N losses into account (Subbarao et al., 2006). However, various N management strategies can reduce the amount of N losses whilst these reductions may not be taken into account by decision support tools. It would be good if the use of these strategies could be described in the model set-up stages so that appropriate N rate recommendations are made.

Linking crop and soil diagnostic measurements with dynamic models of crop growth and development and soil N cycling in computer-based decision support tools should also increase the site specificity of decision-making. Where possible decision support systems should allow for the integration of site specific measurements (Stockdale et al., 1997).
3.5 Best management practices (BMP) for N fertilizer use on crop and pasture land

Introduction

Matching timing of N availability with crop demand is fundamental in reducing N losses in annual cropping systems. As well as selecting the most appropriate rate of N fertilizer, farmers can modify the type, placement and timing of fertilizer applications to reduce N losses and increase fertilizer N use efficiency i.e. the percentage of fertilizer N recovered in the aboveground crop biomass during the growing season.

Crops can take up N from both mineral N forms: \( \text{NH}_4^+ \) and \( \text{NO}_3^- \). A range of N containing fertilizers is used routinely; the most common are urea (c. 40% of fertilizer N applied) and ammonium nitrate (25%). Both undergo microbial conversion in most soils, so that urea is rapidly hydrolyzed and \( \text{NH}_4^- \)-N is rapidly nitrified; consequently within 2-3 weeks of application most mineral N found in the soil is \( \text{NO}_3^- \). In very alkaline soils, \( \text{NH}_4^+ \) can be rapidly lost by volatilization and so fertilizers such as \( \text{CaNO}_3 \) may be preferred to prevent immediate losses.

Optimum N rates can vary significantly within fields (Zebarth et al., 2009). Despite this knowledge, uniform N applications are still the norm, which may result in either over- or under-fertilization in different parts of the field (Fiez et al. 1994b; Kitchen et al 1995; Vetsch et al. 1995, cited by Zebarth et al., 2009). Additionally precision farming or site-specific N management strategies aim to match N supply to crop N demand in space as well as in time (Zebarth et al., 2009).

Slow- and controlled-release fertilizer (SCF) refers to a type of N fertilizer that delays or extends its N availability for plant uptake compared to ‘normal available N fertilizers’ (Subbarao et al., 2006). Often, SCFs are composed of conventional soluble fertilizer materials that have been given a protective coating or encapsulation which controls water entry and rate of dissolution. This allows the release of nutrients to be more synchronized with plant uptake (Fujita et al. 1992 cited by Subbarao et al., 2006). SCFs reduce nitrification as the slow release of N and enhanced competition with the growing crop reduces the availability of \( \text{NH}_4^+ \)-N to nitrifiers.

Nitrification inhibitors (NI) delay the oxidation of \( \text{NH}_4^+ \) by inhibiting nitrifier activity in the soil (Subbarao et al., 2006). Reducing the rate of transformation of \( \text{NH}_4^+ \) into \( \text{NO}_3^- \) allows more time for plant absorption. During the early stages of crop growth, it also allows more time for the crop to reach the stage of rapid growth at which point it can better absorb \( \text{NO}_3^- \) as well as water which reduces the \( \text{NO}_3^- \) leaching potential (Subbarao et al., 2006). Currently nitrapyrin and dicyandiamide (DCD) are the only NIs available that have gained considerable practical and commercial importance in the agricultural and horticultural industry. 3,4-dimethyl pyrazole phosphate (DMPP) has recently been recommended for large-scale adoption in Europe (Subbarao et al., 2006).

Effectiveness

Ideally N applications should be made in multiple small does when plant demand is greatest. The efficiency with which N is used in vegetable crops can be greatly increased through the use of starter fertilizer and banding (Costigan 1989; Stone et al., 1999 – quoted by Goulding 2000). Splitting applications of fertilizer is usually very effective in allowing crop N uptake and minimizing leaching.
loss, especially in soils that are easily prone to leaching (Johnson et al. 1996, 1997; Webb et al. 1997a all quoted by Goulding 2000) or where fertiliser needs to be applied where rainfall is frequent (Powlson et al., 1992; Goss et al., 1993; both quoted by Goulding 2000). Even where leaching risks are not significant during the growing season, an advantage of splitting N applications is that it allows the farmer to identify the sites or treatments that would benefit from additional N fertilization and any second N application to be modified using a diagnostic test at an earlier crop growth stage (Arregui et al., 2006; Arregui & Quemada, 2006).

Jayasundara et al. (2007) showed that under best management practices with fertilizer N rates based on the $N_{\text{min}}$ test and fertilizer N supply synchronized with crop N demand, the fertilizer N uptake efficiency was 58-65% compared to 24-45% under conventional management, without sacrificing yields. However inappropriate split applications may increase leaching risks. For example, farmers are now advised that in temperate climates, winter cereal crops should not be N fertilized in autumn and winter oilseed rape should only receive N fertilizer in autumn if straw has been incorporated previously (Goulding 2000). In a Mediterranean climate, split applications did not reduce cumulative N leaching significantly compared with a single N application with and without a nitrification inhibitor; fertilization occurred at growth stage 25 when crop N uptake was high and leaching minimal (Arregui and Quemada 2006).

Nitrapyrin, DCD and DMPP have been shown to improve N recovery and significantly increase the economic yields of a range of different crops (namely winter wheat, corn, rice, grain sorghum, potato, sugar-beet and cotton; see review by Prasad and Power 1995; cited by Subbarao et al. 2006). However, the use of NIs may not be suitable for some crops or cultivars that do not grow well at high levels of NH$_4^+$ (Sahrawat 1980b, cited by Subbarao et al. 2006). Weiske et al. (2001; cited by Subbarao et al. 2006) made direct measurements of NO$_3^-$ losses and showed that leaching was reduced and grain yields were increased, where DMPP was applied in the field.

The use of polymer-coated urea substantially reduced N losses in the field (largely by reducing volatilization), which in turn reduced the fertilizer requirement by about 40% in comparison to normal fertilizers (Shoji and Kanno 1994, cited by Subbarao et al. 2006).

Factors affecting implementation

Nitrapyrin, DCD and DMPP are persistent at low temperatures ($\leq 5^\circ$C), with their effectiveness being able to last up to six months (Subbarao et al. 2006). In addition, nitrifier activity is usually lower at lower temperatures, which then makes it easier for inhibitors to control nitrification. Hence they can be used more effectively with autumn or winter N fertilizer applications. In regions of Europe where winters are cold and during which period most of the leaching occurs, this should not be problematic. It may however reduce the feasibility of a strategy using NIs in Mediterranean regions with mild wet winters and warm to hot dry summers where irrigation may lead to leaching losses.

In general, nitrification inhibitors are more effective in light-textured soils, soils low in organic matter (ca $\leq 1\%$) and at low temperatures (Sahrawat 1980b, Sahrawat and Kenney 1985; cited by Subbarao et al. 2006). Sorption of NIs onto SOM reduces their mobility and bioactivity and thus their effectiveness (Keeney 1986, cited by Subbarao et al. 2006).
In alkaline soils, the use of NIs can lead to accumulation of NH$_4^+$-N, which may lead to increased losses via ammonia volatilization, especially from surface applied N (Sahrawat 1989; cited by Subbarao et al. 2006). In these specific cases, nitrification can actually lead to N losses; hence the use of NIs will not be suitable in these situations.

Subbarao et al. (2006) showed that where the cost of nitrapyrin or DCD is about 25-30% of the cost of conventional N fertilizer, the use of NIs can be economically profitable if the long-term average N losses exceed 40 to 50 kg N ha$^{-1}$.

Lack of knowledge on the spatial variability of soil N supply during time, misuse of information and lack of qualified services has limited the broader adoption of site-specific N management strategies within precision farming (Baxter et al. 2003; Robert 2002; cited by Zebarth et al. 2009).

**Further research needed**

Plants can regulate nitrification through specific mechanisms such as the release of various compounds as root exudates (Baath et al. 1978; Van Veen et al. 1989; Liljeroth et al. 1990; Parmelee et al. 1993; cited by Subbarao et al. 2006). If these mechanisms are better understood and characterized, they could potentially be transferred to major crops (Subbarao et al. 2006).

Besides taking account of the spatial variability within fields, Kirchmann et al. (2002) also recommended the development of approaches to take account of spatial variability on a larger scale and identify fields from where losses are likely to be largest. Management practices to reduce N loss are most effective on the most sensitive fields or parts of a field. Kirchmann et al. (2002) suggest that N-index assessment systems could be developed (like the P-index assessment systems) where losses by leaching are likely to be the largest proportion of total loss. This could be used to identify the probable causes of the leaching and thus help in finding suitable countermeasures specifically for that “hot spot” (Kirchmann et al. 2002).

The effectiveness of site-specific N management strategies could be improved by the development of through the use of a holistic approaches, which consider both soil spatial variability (e.g. through management zones) as well as spatial variation in crop N sufficiency e.g. through an optically based measure of canopy chlorophyll concentration (Zebarth et al. 2009).

Raisic and Weersink (2008) have carried out some detailed work studying farmer’s responses to N fertilizer rate predictions to identify why over-fertilization remains relatively common. Such work should be developed to give a strong underpinning to the development of appropriate fertilizer recommendation systems.
3.6 Best management practices (BMP) for manure use on crop and pasture land

Introduction

Because the rate at which manure is applied on individual fields is determined to a large extent by the number of animal units on the farm, the number is regulated in many European countries. In Sweden, for example, a stocking density of 1.6 large animal units per ha is allowed (Kirchmann et al. 2002). At field scale, nitrate leaching losses from manure applications depend on:

1. Manure application rates
2. The readily available N content of the manures i.e. the proportion of the total N in the ammonium or nitrate form;
3. Use of nitrification inhibitors
4. The timing of application and subsequent drainage volumes (Chambers et al., 2000)

Effectiveness

The testing of manures to determine their nutrient content, particularly the levels of ammonium-N, is essential for accurate calculation of the correct amount of manure to apply. The use of on-farm N meters (e.g. Quantofix or Agros) for analysis of slurry and manure N contents, are valuable tools to improve the accuracy of application rates. While no studies have directly linked manure testing with nitrate leaching, it has been shown very clearly that high rates of manure application result in greater risks of N losses to water and to the atmosphere (Feng et al., 2005; Gilley et al., 2008). Simple models are available to provide guidance to farmers on the total and available N contents of manures (e.g. MANNER provides a quantitative measure of the fertilizer N replacement value of manure applications; Chambers et al., 1999b; cited by Dampney et al., 2000).

It is also crucial to monitor the N supplying potential of the soil. Repeated applications of organic sources of nutrients will increase the mineralization potential of the soil. Feng et al. (2005) predicted that after a few years of regular manure applications, a “steady state” would be reached where an equivalent amount of nitrogen applied in the organic form will be mineralized during a year. Goulding et al. (2000) showed very high losses of N by leaching where manures had been applied to arable land for over 100 years; in part as a result of the much higher rates of N mineralization.

Efficiency of N use from manures by the crop can be improved through optimum timing and placement of manure applications. For example, van Es et al. (2006) compared different strategies for application of liquid dairy manure to maize and grassland and found higher rates of nitrate leaching when manure was applied in the autumn, but no differences in leaching when spring applications were split or applied at once. The rationale for splitting manure applications is the same as for splitting applications of fertilizer N: split applications allow the N source to be applied nearer to the time when the growing crop will be able to use it. This approach is confounded somewhat when using organic manures, since all of the N in the manures is not readily available. It
is likely that split applications of manures containing high amounts of readily available N will improve efficiency of N use and reduce N leaching. For example, Trindade et al. (2009) studied strategies for applying slurry in a double-cropping system of irrigated maize silage and Italian ryegrass in Portugal. They compared a single application of slurry before maize planting in May, with two split applications before maize planting in May and before ryegrass planting in October, and three split applications before maize planting in May, before ryegrass planting in October and a topdressing of the ryegrass in February. Levels of residual nitrate in the soil to a depth of 1 m in October were highest when the application was not split (122 kg \(\text{NO}_3^-\text{N ha}^{-1}\)) and lowest for the two split applications (92 kg \(\text{NO}_3^-\text{N ha}^{-1}\)). However, it should be noted that the total amounts of N applied for each treatment were not equivalent. The control received a total of 411 kg N ha\(^{-1}\), the May/October split application received a total of 536 kg N ha\(^{-1}\) and the May/October/February split received a total of 478 kg N ha\(^{-1}\). A simulation study of strategies to manage liquid dairy waste in California concluded that frequent low applications resulted in less leaching of N and optimized crop yields compared to less frequent higher applications (Feng et al., 2005).

Banded application or injection is often recommended for manures with high \(\text{NH}_4^+\) contents to reduce volatilization losses. Sistani et al. (2009) found significant reductions in N losses in surface runoff when broiler litter was banded below the soil surface compared with surface application to permanent pastures. However, they did not report N losses due to leaching or denitrification, which may be enhanced when manure is injected into the soil. Vallejo et al. (2005b) studied the effects of application method (injection versus surface application) and the use of the nitrification inhibitor dicyandiamide (DCD), on gaseous losses (\(\text{N}_2\text{O}\) and NO) and losses from leaching. Surface application of slurry resulted in significantly lower losses of N due to leaching and slightly lower losses of nitrous oxide (probably from nitrification of manure-ammonium) than the injected treatment; however, this was likely due to enhanced ammonia volatilization. They also found that a combination of manure injection with DCD reduced N leaching losses to 0.11 g \(\text{NO}_3^-\text{m}^2\) compared with 0.78 g \(\text{NO}_3^-\text{m}^2\) in the injected treatment. These results demonstrate the increased challenges of balancing the need to conserve N for crop growth, with the need to minimize environmental damage when using manure N rather than synthetic fertilizers.

Enhancement of N losses from applied manure by denitrification has been proposed as one strategy to reduce the risk of N leaching from high rates of manure and/or effluent (Schipper & McGill, 2008). This was tested in a study that used “denitrification layers” of organic matter installed at a depth of 30 cm below test plots. A total of 798 kg N ha\(^{-1}\) was applied in effluent. Leaching was reduced from 296 kg N ha\(^{-1}\) in the control plots to 238 kg N ha\(^{-1}\) in the treated plots, but significant amounts of organic N leached past the organic layer. It was concluded that this approach would not be practical on a large scale. In addition, the risk of production of greenhouse gases (e.g. \(\text{N}_2\text{O}\)) would also be high for this method.

Composting of manure has been suggested as a strategy that can be used to stabilize the N in the manure and reduce losses at application due to leaching, ammonia volatilization or denitrification. However as highlighted earlier (Section 3.2) significant losses occur during the composting process. Basso and Ritchie (2005) found that the use of composted dairy manure in a maize-alfalfa rotation reduced the cumulative leaching over 6 years to 146 kg N ha\(^{-1}\) from 341 kg N ha\(^{-1}\) when fresh manure
was used compared to when manure was composted. Hepperly et al. (2009) showed that the use of compost not only reduced losses of N to surface and groundwater, but also improved soil nutrient levels and provided residual soil fertility for the wheat crop. The use of compost may also improve soil physical properties which can impact on runoff volume. For example, Evanylo et al. (2008) found that runoff volume from compost-amended vegetable plots was much lower compared with manure, fertilizer or control treatments. This was coupled with greater water infiltration in compost plots, but did not result in any differences in N leaching among the treatments. In contrast, a study in Arizona on an alfalfa crop did not indicate any differences in the potential for nitrate leaching (indicated by levels of nitrate-N in the soil profile to 150 cm) after repeated applications of equivalent amounts of N as composted dairy manure or fresh dairy manure over 1.5 years (Martin et al., 2006). Ferguson et al. (2005) applied the same rates of total N from manure (assuming that 35% of the total N would be available in the year of application) and compost (assuming 25% N availability) as N sources for maize. In some years NO$_3^-$-N accumulated in the root zone (0-1.5 m) and leached into the vadose zone (1.5-3 m) for the compost treatment, suggesting that more than the assumed 25% total N was mineralizing in these treatments. Evanylo et al. (2008) also highlighted nitrate leaching risks associated with the use of composted manure sources due to mineralization of N that is not synchronous with crop demand, and resulting excess levels of nitrate in the root zone late in the season when it can pose a leaching risk.

Ferguson et al. (2005) showed relatively low soil NO$_3^-$-N concentrations where both fresh manure and compost were applied, indicating that the procedure used to determine manure application rates effectively minimized the potential for substantial NO$_3^-$-N leaching out of the root zone. BMP for manure use are complex and must be site and system specific because of the tradeoffs between the conservation of one N form and concomitant increases in other N losses (Oenema et al., 2009). For example, manure injection into soil to reduce ammonia emissions and improve air quality may increase both nitrate leaching (which reduces ground water quality) and denitrification (which increases greenhouse gas formation).

Factors affecting implementation

Farmer perception of manures as a waste is slowly being displaced by farmer awareness of their value as a resource (Schroder, 2005). However, much more work is needed to provide information and tools for farmers to assess the fertilizer value of on-farm manures.

Chambers et al. (2000) conducted a study of on-farm performance of manure spreaders and found that performance of most spreaders was poor with coefficients of variation for lateral spread uniformity as high as 50% (Chambers et al., 2000). Correct estimation of the amounts of manure applied by the spreader is also essential. Manel and Slates (2003) surveyed 101 farmers and found that when using a visual estimation, 50% of them dramatically underestimated the amounts of manure that had been applied. Accurate calibration of manure (and fertilizer) spreaders is a key component of best management practice for manure use on land (Goulding 2000). Fairly simple calibration tests and adjustments can be made to spreaders (Smith & Baldwin, 1999); farmer understanding and engagement is often the limiting factor.
3.7 Management of crop residues

Introduction

Mineralization from crop residues after harvest can be a large contributor to soil nitrate formation in autumn (Kirchmann et al. 2002). Post-harvest studies suggest that there is a predominance of soil-derived nitrate, thought to amount to 90% or more, in the soil profile in autumn (Bergstrom 1987; Macdonald et al. 1989; cited by Kirchmann et al. 2002). However, organic materials with high C:N ratios, such as straw, tend to cause N immobilization during degradation. The immobilized N is released in subsequent years (Powlson et al. 1987 quoted by Goulding 2000). Hence, soil incorporation of crop residues may have a range of impacts on N leaching depending on the amounts and decomposability (C:N ratio) of the crop residue materials. According to Kirchmann et al. (2002), “one tonne of straw containing 0.4% N can bind 9 kg of inorganic N when it decomposes for one year in the field” and hence straw incorporation may help to reduce overwinter N leaching losses (Powlson et al. 1985, cited by Nicholson et al., 1997). In contrast, the ploughing of grazed grassland is often followed by a large increase in the mineralization of N and the risk of N leaching, due to the considerable build-up of organic N with relatively high decomposability in grassland systems (Eriksen et al. 2004).

Effectiveness

Incorporation of high C:N straw residues after crop harvest may be one strategy to reduce late autumn and winter leaching risks in temperate regions of Europe by promoting immobilization of soil mineral N. Experiments done throughout the UK have confirmed that straw incorporation without any addition of N fertilizer is effective and does not retard growth of subsequent crops (Lord & Shepherd 1996; Nicholson et al. 1997, Glendining & Smith 1999 – all cited by Goulding 2000). Nicholson et al. (1997) researched how different straw disposal techniques (burning or chopping before incorporation) affected soil mineral nitrogen and the potential for nitrate leaching losses in two light textured soils in England (loamy sand at Gleadthorpe and sandy loam at Morley). No consistent effects on autumn soil mineral N were found but cumulative nitrate-N leached was reduced slightly on autumn fertilized plots in one year when drainage volumes were above average. Jarvis et al. (1989; cited by Nicholson et. al 1997) found that straw incorporation reduced nitrate leaching during the growth of winter cereals in loamy sand and clay soils when compared to burning. However, in the studies of Catt et al. (1992; cited by Nicholson et al. 1997) and Davies et al (1996) the overall effects of straw incorporation were small and inconsistent. The variability in the effects of straw incorporation on leaching losses could be a result of differences in the actual C:N ratio due to growing conditions in the crop.

Re-seeding of permanent swards Incorporation of N-rich crop residues will lead to a dramatic increase in soil mineral N, if this takes place in autumn at the beginning of the leaching period leaching risk will be increased very significantly (Möller et al. 2008; Thomsen 2005). Similarly there is a relatively high risk of nitrate leaching during the period immediately after re-seeding of a
permanent sward because of the large amounts of N released by mineralization. The amount of inorganic N released in this process will often exceed the need of the subsequent crop (Frances 1995; Eriksen et al. 1999; cited by Eriksen et al. 2004) and so leaching risk will be very significantly increased even where cultivation takes place immediately before the main growing season. Scholefield et al. (2003) surmised that there would have been a large quantity of nitrate leached from the re-seeded swards in the first winter after cultivation but this was not assessed. To reduce this risk, good management practices following grass cultivation, such as ploughing in late winter or spring, can help to control the amount of N released. In the long term, the benefits of re-seeding permanent swards may outweigh the N leaching risks, since initially large effects on nitrate leaching can be short-lived, with leaching quantities being lower in reseeded swards compared to undisturbed swards after one year (Scholefield et al. 1993; Jarvis 2000; Eriksen et al. 2004). This may be due to large differences in annual net N mineralization between soils from permanent and reseeded swards (Tyson et al. 1991, cited by Scholefield et al. 2003). Soil property measurements taken five years into the study, revealed that ploughing and reseeding resulted in reduced aeration and organic matter content of the 0-10 cm soil layer, which could be expected to lead to lower mineralization rates (Scholefield et al. 1993). Therefore, reductions in annual rates of N mineralization in reseeded swards may also balance the short-term risk of increased N leaching immediately after sward re-seeding.

**Further research needed**

An understanding of the relationship between the C:N ratio of the crop residues and the level of mineral N in the soil that is required to observe a reduction in nitrate leaching is needed. This implies an improved understanding of the C and N dynamics in soils during the winter leaching period and how these dynamics can be managed to maximize N immobilization.

Optimization of tillage and re-seeding operations in renovated swards is needed to minimize the risk of leaching during these operations. Also, further research into strategies to renovate swards without cultivation is needed.
3.8 Modifying crop rotations to minimize N losses to water

Introduction

Crop rotations (the sequence and type of crops grown on a piece of land over time) are already used by farmers to meet a number of management goals including:

1. Diversification of crop types on the farm to minimize economic risk;
2. Diversification of crop types in space and time to combat the risk from pests (insects and disease);
3. Provision of a range of feed types to on-farm livestock e.g. forages and cereals, and;
4. Provision of ecological services including biodiversity, soil conservation, water protection.

There are a number of ways that crop rotations can be modified specifically to minimize N losses to water. These include the use of catch crops, legumes, and deep-rooted crops. Additionally the highest-yielding variety of all crops within the rotation should be chosen to maximize the use of the available N whilst also maintaining quality to reduce N loss (Goulding 2000).

Leguminous crops are often introduced into crop rotations to provide an alternative supply of N to the farming system. Leguminous crops fix N from the atmosphere and convert it to organic N. Through good rotational design and tillage management this source of organic N, when incorporated into the soil, can become part of the soil’s organic N pool and eventually contribute to the soil N supply for subsequent crops. Good use of legumes in crop rotations can therefore reduce the farm’s reliance on purchased, soluble N fertilizers. However, achieving synchrony between crop demand and N release from catch crop or legume residues is more difficult than achieving synchrony where synthetic N fertilizers are used (Robertson & Vitousek, 2009).

Cover crops include both leguminous and non-leguminous species and are planted so that agricultural land does not lie fallow and bare during the non-growing season (Herrera & Liedgens, 2009). Cover crops can interfere with cash crop planting (Tonitto et al., 2006), hence careful rotation management is necessary. A potential disadvantage with the use of legume cover crops is that crop rotation needs to be modified sometimes to permit sufficient time for N-fixation (Tonitto et al. 2006). Cover crops are deliberately planted to reduce runoff and erosion, to reduce drainage and capture “surplus” nutrients from the topsoil and, where weather permits, to provide an N input through N fixation. Cover crops deliberately grown to increase the capture of surplus inorganic N left in the soil after harvest of the main crop as well as N released through on-going mineralization are often known as catch crops (Tonitto et al., 2006; Jayasundara et al., 2007). If cover crops establish successfully they can potentially reduce losses of N via leaching or runoff during the non-growing season (Schröder et al., 1996). Other benefits related to the use of cover crops are:

1. The possibility of increased bio-availability of mineral-derived nutrients, most notably P (Vance et al. 2003 cited by Tonitto et al. 2006).
2. Tapping a greater proportion of the productive capacity of an agroecosystem by avoiding periods of bare fallow.
3. The possibility of enhanced disease suppression (Abawi and Widmer 2000; cited by Tonitto et al. 2006), reduced weed competition and herbicide requirements (Gallandt et al. 1999; cited by Tonitto et al. 2006), and better conditions for beneficial arthropod communities.
that permit reductions in pesticide applications (Lewis et al. 1997; cited by Tonitto et al. 2006).

Catch crop material and legume crop residues (both green manures and cash crops) are often incorporated into the soil to transfer soil inorganic N captured to the next season’s crop, and potentially reduce the fertilizer need of subsequent crops (Schroder et al 1996). The use of catch crops has been found to have variable effects on subsequent crops (Tonitto et al. 2006). This might be due to the effect catch crops have on soil fertility over the short and long term (Constantin et al. 2010). Catch crops have been shown to increase soil organic N accumulation (Kuo & Jellum, 2000); however, yields of crops grown after a catch crop may decrease or remain the same in the short term. Over a longer period however, it is expected that repeated use of catch crops will benefit subsequent crops through soil organic N accumulation (Kuo & Jellum 2000). Legume-based crop rotations or pasture systems have reduced fertilizer inputs compared with systems where no legumes are integrated which may also have benefits in reducing dependence on fossil fuels (Tonitto et al. 2006).

As well as incorporating catch crops and leguminous crops into the crop rotation, land managers have the opportunity to design their crop rotations to take advantage of the potential of deep rooted crops to capture N that has leached deep into the soil profile. By alternating deep-rooted crops with shallow-rooted crops, farmers can recover mineral N in one year that may have been leached below the rooting zone in previous years.

**Effectiveness**

Catch crops have been found to be very efficient in reducing N leaching in the season they are grown. For example, in three long term experiments (ranging from 13 to 17 years) in Northern France, Constantin et al. (2010) found that catch crops reduced N leaching between 36 to 62% compared to bare fallow in the autumn. In rotation using very high rates of N fertilizer, the effectiveness of catch crops in limiting N leaching is reduced. Schroder et al (1996) found indications that catch crops may have also stimulated N loss via denitrification or immobilization at high N input levels. To obtain insight into the effect catch crops had on subsequent cash crop yields and N retention, Tonitto et al. (2006) performed a meta-analysis which showed a consistent positive effect of catch crops on N leaching in the season they were grown. A non-legume cover crop rotation reduced leaching by 70% on average compared with bare fallow (N = 69; Tonitto et al. 2006). The use of leguminous over-wintering cover crops also reduced N leaching. Nitrate leaching was reduced significantly by 40% on average when legume cover crops were used compared to conventional systems (N = 18). Post-harvest soil nitrate status (as a measure of potential N loss) was similar in both legume cover crop and bare-fallow rotations, which indicated that N leaching reductions were largely caused by the avoidance of bare fallow periods (Tonitto et al. 2006).

The factors affecting the impact of the incorporation of cover crop residues and their direct impacts on leaching are the same as for any other crop residues (Section 3.7). However, as the incorporation of cover crops is a deliberate management intervention, it is important to be able to assess the medium to long term impacts on cash crop yields and on N leaching. Catch crops have been shown to have highly variable effects on N availability for succeeding crops, ranging from negative to
positive (Berntsen et al., 2006). For example in a study in which a ryegrass catch crop was grown repeatedly for three years, although nitrate leaching losses were reduced, no effects were detected on the subsequent barley crop's uptake of N or its grain yields (Thomsen, 2005). Constantin et al. (2010) showed that after 7 years of use of non-leguminous cover crops some small yield increases were seen. Tonitto et al. (2006) used a meta-analysis to study the effect of preceding legume cover crops on yield. On average, legume-fertilized crops had 10% lower yields compared to conventional systems \((N = 206)\), unless legume biomass was \(\geq 110 \text{ kg N ha}^{-1}\), in which case yields were not significantly different between the two systems. In cropping systems that include legumes, a higher proportion of N obtained by crops grown following the legume incorporation is derived from SOM (80% compared to 40-60% in crops receiving inorganic N fertilizer (Tonitto et al 2006). Constantin et al. (2010) calculated that the N storage in SOM was increased by approximately 10-24 kg ha\(^{-1}\) yr\(^{-1}\) at the sites studied. Berntsen et al. (2006) used a simulation model to study long-term effects and indicated that N leaching can increase in the long term (after approximately 10 years) in systems which continuously use catch crops, if fertilizer N is not reduced or the crop rotation is not changed. This is a result of the impact of increases in organic matter and hence ultimately rates of mineralization. When considering the 75 years following a 25-year rotation using catch crops, between 33 and 47% of the built-up soil organic N was predicted to be lost as leaching (depending on soil type, climate and soil fertility; Berntsen et al. 2006). Approximately 35% of the N retained due to reduced N leaching was recovered in rotation meaning that N fertilization could be reduced for crops with a pre-history of catch crop use, without yield reductions.

Thorup-Kristensen et al. (2006) studied the effects of incorporating deep rooted cash and catch crops into crop rotations along with shallow-rooted crops. They compared very different rooting depths from only 0.5 m (leek), to similar to 1.0 m (ryegrass and barley), 1.5 m (red beet), 2.0 m (fodder radish and white cabbage) and more than 2.5 m by the chicory catch crop. They found that a significant portion of mineral N was retained within the soil profile from one year to the next, and that by alternating deep rooted crops with shallow-rooted crops, this N could be recaptured. White cabbage reduced the levels of inorganic N in soils deeper than 1 m by 113 kg N ha\(^{-1}\) during one season. Catch crops grown in combination with shallow-rooted crops like leek and red beet, were able to reduce levels of mineral N in the >1 m layer by on average 60 kg N ha\(^{-1}\). The authors concluded that:

> it is possible to design crop rotations with improved nitrogen use efficiency by using the differences in crop rooting patterns; deep-rooted crops or catch crops can be used to recover Ninorg leached after previous crops, and catch crops can be grown before shallow-rooted crops to lift the deep Ninorg UP to layers where these crops have their roots.

### Factors affecting implementation

Catch crop planting and type should be appropriately matched to available niches in time and space to avoid conflicts between catch crops and cash crop planting and possible yield reductions.

\[N\text{-TOOLBOX D1.1}\]
Clover content is difficult to maintain in grass-clover swards. Eriksen et al. (2004) showed that dry matter production was maintained in older, cut fertilized grass swards whilst it decreased by almost 50% in cut grass-clover swards in the experiment researched. Maintaining production levels for livestock systems is more difficult in legume-based pastures.

Availability of cover crop seeds that are adapted to local environmental conditions can be limited.

The purchase of cover crop seeds and establishment will be an additional cost to the producer that may not be economical at current values for fertilizer N.

Future research

In the case of legume-fertilized cropping systems, the interactions between SOM pools, N-fixation and microbially-mediated processes must be characterized to optimize yields and N balance. A system is needed to discern when supplemental N fertilizer is needed in conjunction with legume inputs to avoid N inputs being insufficient for crop N demand. Likewise, the possibility of surplus N addition in legume-based systems should also be taken into consideration (Tonitto et al. 2006). This all implies a need for improved systems (integrating measurement and models) to predict the amount and timing of soil N supply from a variety of soil organic matter pools of varying quality.

Regionally-specific cropping systems should be designed that take local ecosystem and socio-cultural conditions into account (Tonitto et al. 2006). This should include a focus on development of cover crop varieties that are compatible with dominant crop rotations and climatic constraints (Frye et al. 1988; cited by (Tonitto et al., 2006).

Although cover crops are mostly ploughed into the soil, an alternative management option is to harvest the aboveground biomass for feed or as substrate for a biogas digester (Möller et al. (2008). More research is needed to inform a full Life Cycle Analysis of the impacts of management approaches for cover crops.
3.9 Modifying tillage practices to reduce N losses to water

Introduction

Tillage aerates the zone of soil disturbance and so increases aerobic microbial activity which leads to increased oxidation of soil organic matter and mineralization of organic N present in the soil (Shepherd et al., 1993; Dinnes et al., 2002). Thus tillage practices and their timing interact with N leaching losses. Relatively small impacts on N leaching are achieved through changes in types and timing of tillage when compared to other techniques (Bakhsh & Kanwar, 2007; Henke et al., 2008; Constantin et al., 2010).

Conservation tillage or minimum tillage is associated with many benefits such as: the reduction of erosion; minimization of damage to soil invertebrates; development of better soil structure, and; reduction in the use of fossil fuels during seedbed preparation. In addition, spilled seeds that germinate after harvest in minimum till systems can form a relatively dense ground cover and can thus act as a catch crop (Kirchmann et al., 2002).

Effectiveness

Cultivation of the soil generally accelerates the post-harvest accumulation of nitrate (Shepherd et al., 1993; Davies et al., 1996). Delaying cultivation until just before the next crop can delay the release of nitrate and hence decrease the N leaching risk as it allows subsequent crops to assimilate N as they become established (Shepherd et al., 1993; Dinnes et al., 2002). However, it is not advised to delay cultivation for too long after crop harvest where autumn drilling is taking place; crops sown after mid-October take up little N before ceasing growth overwinter and the N mineralized by cultivation cannot be taken up by the crop (Shepherd et al. 1993). As mineralization depends on temperature and rainfall (Dinnes et al. 2002), changes in the timing of tillage may not always bring about the desired effect.

A study in Denmark demonstrated the effects of both catch crops and reduced tillage. At a site with a coarse sandy soil the period in which cultivation took place did not affect the average annual leaching unless a catch crop was included (Hansen & Djurhuus, 1997). Where this was the case, the average annual leaching over a five year period was 31% less in the plots ploughed in spring (29 kg NO₃-N ha⁻¹ year⁻¹) compared to those ploughed in autumn (42 kg NO₃-N ha⁻¹ year⁻¹). In contrast, the timing of tillage affected the average annual leaching in all treatments in a sandy loam soil. Average annual N leaching was lower in the treatments that were ploughed in spring compared to late autumn (mid-November) ploughing. Average annual leaching over a four year period was reduced by 25% in plots that were ploughed in spring compared to plots ploughed in late autumn (from 65 to 49 kg NO₃-N ha⁻¹ year⁻¹). Similarly, in treatments where stubble was cultivated in autumn, the average annual leaching over a four year period was reduced by 17% (from 76 to 63 kg NO₃-N ha⁻¹ year⁻¹) if ploughing took place in spring instead of late autumn. In treatments where a catch crop (ryegrass) was grown, average annual N leaching was reduced by 38% (from 53 to 33 kg NO₃-N ha⁻¹ year⁻¹) if crops were cultivated in spring rather than late autumn (Hansen and Djurhuus, 1997). Their study took place in Denmark, under temperate coastal climate conditions. It was thought that the
difference between autumn and spring cultivation would have been more if ploughing in autumn had taken place earlier (Hansen and Djurhuus, 1997).

Reducing overall tillage intensity may be an option for farmers with suitable soil structure and aeration status which depend on soil texture and climate (Rochette, 2008). Poorly drained soils (these tend to be clayey) are usually unsuitable for zero tillage (Rochette, 2008). Use of no-till practices decreased NO$_3$-N leaching rate when compared with conventional tillage, however, this was accompanied by increased N$_2$O emissions (Farahbakhshazad et al., 2008; Mkhabela et al., 2008; Constantin et al., 2010). In a review of 25 field studies comparing conventional tillage with no-till, it was seen that no-till generally increased N$_2$O emissions in soils that were estimated to be poorly-aerated (Rochette 2008). Stoddard et al. (2005) compared two types of conservation tillage, namely no tillage and chisel ploughing followed by secondary discing (CD). Tillage had few statistically significant impacts on flow-weighted NO$_3$-N concentrations and hence on expected leaching losses.

No tillage has also been linked to greater mineralization and increased amounts of organic matter (Doran 1980; Blevins et al. 1983, cited by Stoddard et al. 2005) as well as enhanced macropore flow (McMahon and Thomas, 1976; Tyler and Thomas, 1977; Shipitalo and Edwards, 1993, cited by Stoddard et al., 2005). Hence, in the medium-long term minimum tillage practices may increase leaching risks as macropore channels in the topsoil, which can act as preferential flow paths, remain intact. Applied N fertilizer may bypass the root zone and reach deeper soil layers (Kirchmann et al., 2002). Thus, extra attention should be paid to the timing of N fertilizer applications in reduced tillage systems.

*Factors affecting implementation*

Account should be taken of local soil and climate conditions when deciding on management practices to be combined with no-till (Farahbakhshazad et al., 2008). Field studies in Midwestern croplands of USA as well as model simulations have indicated that the impacts of no-till are highly variable in space and time due to companion management practices (e.g. crop rotation), as well as the climatic and soil conditions (Farahbakhshazad et al., 2008).
**3.10 Strategies for irrigated cropland**

**Introduction**

Systems of crop production that use irrigation can be considered a special case when identifying strategies to reduce N losses to water. In these systems soil water contents are closely managed for optimum crop production, and many of the strategies to reduce the movement of N into water involve the management of irrigation water. In addition, many of the strategies that apply in non-irrigated production systems, e.g. accurate estimation of N fertilizer rates and the use of in-field strategies to improve efficiency of N cycling, also apply in irrigated systems. Therefore, the need to consider soil water conditions and the management of irrigation water in addition to the usual considerations required to optimize N management, adds another level of complexity to the problem of managing N losses to water from irrigated systems.

Irrigated production systems tend to be located in regions where the climate is characterized by a warmer dry period (during which a crop is produced and irrigation is used) followed by a cooler wet period (during which the land may lie fallow, or be planted to a catch crop or a cool season cash crop). This divides the year into two periods (irrigated and non-irrigated), during which different strategies to reduce N losses to water can be employed.

A key approach to reducing N leaching during the irrigated period of crop production is to **ensure that the wetted volume of the soil does not exceed the boundaries of the root zone.** As the nitrate tends to accumulate towards the boundary of the wetted volume, the use of irrigation strategies that limit the wetted volume in the root zone may improve water and nitrogen fertilizer use efficiency (NUE), as well as reduce nitrate leaching (Bar-Yosef and Sheikholeslami, 1976; Li et al., 2003 and Singandhupe et al., 2003, quoted in Zotarelli et al., 2009).

A key approach during the non-irrigated period of the year, is to **minimize the downward movement of residual N into groundwater,** by capturing residual N in a catch crop, or by incorporating high C:N ratio residues that may immobilize mineral N.

A major factor resulting in N leaching from irrigated systems is the use of excessively high levels of N inputs. Although the science demonstrates that applying excess levels of N to the crop in irrigated systems enhances the risk of N leaching, the practice is still common. A survey of soil-based greenhouse production systems in southern Spain showed that applied fertiliser N was > 1.5 and > 2 times crop N uptake in respectively, 42 and 21% of crops surveyed. Large amounts of manure were applied to the land at the greenhouse construction stage, averaging 3046 kg N ha\(^{-1}\). Periodic manure applications were made in 68% of the greenhouses, with average application rates of 947 kg N ha\(^{-1}\). These applications of manure-N were not accounted for in the fertilizer programmes in 74% of the greenhouses (Thompson et al., 2007).

**Effectiveness**

Similar to non-irrigated crop production systems, **application of the correct N fertilizer rate** is the first line of defense against N losses to water by leaching in irrigated systems. Fonder et al. (2010) demonstrated the importance of **not exceeding recommended fertilizer N rates.** They compared...
three different levels of N fertilization with and without irrigation with wastewater, in a factorial design. The experiment was located in Belgium and the crops studied were: spinach, bean, carrot and broad bean, and winter wheat. They found that:

The nitrogen residues in the soil after harvest were acceptable and regular as long as the fertilisation advice was not exceeded; the maximum fertilisation level tested, 50% higher than the recommendation, systematically left unacceptable nitrogen residues in the soil, harmful for the environment.

Similarly, Zotarelli et al. (2007) demonstrated that applying N rates in excess of standard recommendations increased N leaching by 64, 59, and 32%, respectively, for pepper, tomato, and zucchini crops grown with irrigation in Florida, USA.

Determination of correct N application rates should be based on similar approaches as for non-irrigated crops (see Section 3.4), specifically crop yield and uptake of N, and the N supplying capacity of the soil should be considered. Soil testing and simulation models that include good soil water sub-modules that can simulate irrigation events should be used.

Applications of N fertilizer to irrigated crops can be further optimized using precision farming techniques. Delgado et al. (2005) used GIS, GPS, and modeling technologies to identify and simulate the spatial residual soil NO$_3$-N patterns in a centre pivot irrigated corn (Zea mays L.) field. Using a combination of soil sampling and mapping they were able to identify areas of the field that were low in residual nitrate and also had a low potential productivity. The N Reflectance Index method was used to maximize the synchronization of "in season" N applications with corn N uptake. This increased N use efficiencies and reduced NO$_3$-N leaching losses by 47 percent when compared to traditional practices ($P < 0.0001$).

A much simpler, but still effective method to modify in season N applications is to use hand-held leaf-greenness measuring devices. Technology as simple as a leaf colour chart can be used to determine the need for additional applications of N, and thereby optimize fertilizer recommendations (in both irrigated and non-irrigated systems). Colour charts are used by farmers as a decision making tool that allows them to assess the colour (greenness) of their crop in the field and apply additional N when the colour indicates N deficiency. This simple approach was demonstrated to effectively reduce fertilizer N use by an average of 26% in irrigated rice systems of India, without adversely affecting crop yields (Singh et al., 2007).

In irrigated systems, management of soil water contents can be just as important as N management for controlling the downward movement of nitrates. Gheysari et al. (2009) demonstrated that even at relatively low rates of N fertilization, leaching below 60 cm could occur when irrigation water was applied in excess of crop requirements. Vazquez et al. (2006) studied irrigated tomato production systems in Spain and demonstrated that excessive irrigation during the tomato establishment phase could cause large losses of N.

While applying excess irrigation water can enhance leaching of nitrate below the root zone, enough water should be added for optimum plant growth. This will ensure that the capacity of the crop to take up mineral N from the soil is maximized and N residues in the soil after harvest are minimized.
(Fonder et al., 2010). As well as benefits to plant growth from irrigation, Fonder et al. (2010) found higher N residues in the soil in non-irrigated treatments and suggested that the irrigation improved the “solubility” of N in the soil, allowing it to be better taken up by the plant.

The use of manure as a source of N is common in irrigated systems and many of the same strategies should be implemented as for non-irrigated systems, to minimize losses of manure-N to water. Experiments with pig slurry and N fertilizer in irrigated maize systems in northern Spain demonstrated the importance of accounting for soil mineral N contents when determining N fertilizer needs, and of applying optimal N rates. In this study, annual optimal N rates gave the lowest soil NO$_3^-$-N contents after harvest and the lowest N losses (Berenguer et al., 2008). Injection of pig slurry in combination with the nitrification inhibitor dicyandiamide (DCD) has been shown to reduce nitrate leaching compared to treatments without DCD (Vallejo et al., 2005a). And in irrigated systems using liquid dairy manure as an N source, a winter forage crop following a summer corn crop effectively reduced the leaching of residual soil N following the corn crop (Feng et al., 2005).

Simply reducing the concentrations of N in fertigation systems in greenhouses can reduce nitrate leaching. Munoz et al. (2008) showed that the N concentration in the nutrient solution in a fertigated greenhouse can be reduced from 11 mM (standard management practice) to 7 mM under a daily mean drainage volume of 30%. This finding implies a 70% decrease in nitrate leaching without reducing tomato yield or quality.

Management of irrigation water and N leaching can be improved by substituting drip irrigation systems for furrow irrigation. Converting from furrow-irrigated to drip-irrigated onion production may reduce N fertilizer needs, water inputs, and NO$_3^-$-N leaching potential. Halvorson et al. (2008) compared drip and furrow irrigation of onions at a range of N fertilizer rates. They found that irrigation water use efficiency (IWUE) and N use efficiency (NUE) were higher with the drip system than with the furrow system. In one year residual soil nitrate levels were higher for drip irrigation systems, but this may have reflected increased leaching of nitrates in the furrow irrigation systems. The higher NUE in the drip system suggests that the sum of N losses by all pathways were lower in this system.

Improvements to irrigation water management can result in improvements in fertilizer use efficiency, as demonstrated by Zotarelli et al. (2009) in an irrigated tomato production system in Florida. They investigated soil moisture sensor-based (SMS) irrigation systems coupled with either: a) surface irrigation and fertigation drips both on the soil surface (SUR), or b) use of sub-surface drip irrigation placed 0.15 m below the soil surface and fertigation drips on the soil surface. The SMS systems in both cases were programmed to provide irrigation water in five windows per day, when volumetric water content was 0.10 m$^3$ m$^{-3}$. These two systems were compared with a once daily fixed duration surface applied irrigation treatment and surface fertigation. The authors found that the use of a soil moisture sensor-based irrigation system consistently reduced irrigation water use, volume percolated, and nitrate leaching. Nitrogen use efficiency was significantly higher for both SMS systems compared to the control method.
**Catch crops** were used in the Fonder *et al.* (2010) study as a tool to take up excess mineral N after the main crop harvest. They found that seeding had to be in late summer in order for the catch crop to effectively reduce the risk of N leaching; later seedings did not effectively reduce the N leaching risk.

Nitrification inhibitors, such as dimethylpirazole phosphate (DMPP) can be used in irrigated systems to reduce the rate of $\text{NH}_4^+$-N transformation to $\text{NO}_3^-$-N and hence reduce the risk of $\text{NO}_3^-$-N leaching. Quinones *et al.* (2009) demonstrated that including DMPP in a fertigation system for a citrus orchard increased the levels of $\text{NH}_4^+$-N and reduced $\text{NO}_3^-$-N levels in the top 0-40 cm of soil compared to treatments without DMPP. Diez-Lopez *et al.* (2008) demonstrated that the use of DMPP with urea in an irrigated maize crop reduced nitrate leaching and did not affect yield at equivalent rates of N.

The **use of simulation models** in irrigated systems may be particularly useful for identifying strategies to reduce N losses to water. Prediction of N leaching from irrigated systems requires an understanding of the interactions between various factors: soil water balance, soil N mineralization, and crop N uptake. If a model can accurately predict the dynamics of each of these processes during the growing season, and successfully transfer information among the processes, then it could be a useful tool for investigating the impacts of various management scenarios on N leaching. Doltra and Munoz (2010) compared the crop-based simulation model (EU-Rotate_N) and a widely recognized solute transport model (Hydrus-2D) in a simulated bell pepper–cauliflower–Swiss chard rotation in a sandy loam soil. Predicted values for: soil water content, water draining below 60 cm, nitrate nitrogen ($\text{NO}_3^-$-N) contents in the 0–90 cm soil profile, crop uptake of $\text{NO}_3^-$-N, $\text{NO}_3^-$-N leaching below 60 cm, and crop yield were compared with actual values measured in the field (near Barcelona, NE Spain). The EU-Rotate_N was identified as a potential tool for investigating management scenarios in irrigated systems (values of the mean absolute error (MAE) of predictions were below the average standard deviation of the observations from the field in most cases).

A simulation model was also used by Sophocleous *et al.* (2009) to investigate options for reducing N leaching from wastewater irrigated maize in Kansas, USA. Using their model, they demonstrated that reducing levels of corn N fertilization by more than half to 170 kg ha$^{-1}$ substantially increased N-use efficiency and achieved near-maximum crop yield. Combining such measures with a crop rotation that includes alfalfa was predicted to further reduce the accumulation and downward movement of $\text{NO}_3^-$-N in the soil profile.

The simulation model STICS was used by Jego *et al.* (2008) to investigate scenarios for irrigated potato production in northern Spain. Results from simulations identified the following recommended practices for reductions in nitrate leaching:

1) N-fertilizer should not be applied in autumn before winter crops;
2) crops with low N uptake capacity (e.g. potatoes) should be avoided or should be preceded and followed by nitrogen catch crops or cover crops;
3) the nitrate concentration of irrigation water should be taken into account in calculation of the N-fertilization rate, and
4) N-fertilization must be precisely adjusted in particular for potato crops.
Gallardo et al. (2009) combined the TOMGRO model that simulates N uptake for tomatoes grown in greenhouses in SE Spain with the PrHo model to simulate transpiration of tomato grown in substrate. The improved model was used to calculate N uptake concentrations and drainage NO$_3$-$N$ concentration in free-drainage or “open” substrate greenhouse vegetable production. The authors concluded that the aggregated model was reasonably accurate: simulated values of average NO$_3$-$N$ concentration in drainage obtained with the aggregated model were -7, +18 and +31% of measured values.

Rinaldi et al. (2007) used the CROPGRO simulation model to look at the economic and environmental impacts of a range of irrigation and fertilizer management scenarios in irrigated processing tomatoes in southern Italy. Overall conclusions were that frequent irrigation applications combined with low N rates reduced crop stress and represented the best scenario from both a production and environmental point of view (low N leaching). The economic evaluation indicated no negative effects from reducing irrigation from 75 to 50% of crop available water in the soil. Optimum N application was identified as not less than 200 kg of N ha$^{-1}$. This study also demonstrated the value of using a simulation model as a decision support tool to identify optimum management strategies.

Other simulation models which can be used include NLEAP (Nitrogen Loss and Environmental Assessment Package) which De Paz (1999) used to evaluate irrigated systems with vegetable crops such as potato, cauliflower (Brassica oleracea var. botritys), and onion (Allium cepa L.) grown in a Mediterranean region of Spain. Delgado et al. (2008) expanded NLEAP to assess management options for irrigated cropland from an arid Western US site using a Windows XP version of NLEAP with Geographic Information System (GIS) capabilities (NLEAP-GIS). The model was used to compare reactive N losses from baseline and alternative scenarios. When the difference between the two scenarios (Nitrogen Trading Tool difference in reactive N losses or NTT-DNLR_reac) was positive, this indicated that less reactive N was being lost from the alternative scenario than the baseline. In the scenarios they simulated, alternating deep-rooted (barley) crops with shallow-rooted (potato) crops was investigated, demonstrating the value of the deep-rooted crops as scavengers of N that had moved down in the soil profile beyond the reach of the potato roots. In future systems where improved management of N may be incentivized through trading in N credits, this simulation model may be used to identify the most profitable management strategies for farmers.

“End-of-pipe” solutions may sometimes be necessary in irrigated systems. Constructed wetlands for treatment of runoff water from irrigated pastures have been tested in California. Findings were that if wetlands were well maintained and inflow rates managed, then water quality could be significantly improved with loads of total suspended sediments, nitrate, and Escherichia coli reduced on average by 77, 60, and 68%, respectively, and retention of total N, total P, and soluble reactive P (SRP) of between 35 and 42% of loads entering the reference wetland (Knox et al., 2008).
3.11. Runoff, drainage and wastewater management

Introduction

Housing, manure handling and storage and in-field approaches such as those discussed above may not always be able to reduce nitrate concentrations in water leaving the farm to zero; for example Verloop et al. (2006) showed how an integrated approach to reducing N losses from a dairy farm reduced NO$_3^-$ concentrations in groundwater from 193 mg l$^{-1}$ to 63 mg l$^{-1}$; however, concentrations stabilized at this level which is higher than the EU standard of 55 mg l$^{-1}$. In such a situation, further reduction of diffuse pollution from farms might be achieved though so called “end-of-pipe” solutions which seek to reduce the amount of water leaving the farm or to treat water leaving the farm in surface water streams, ditches or in drains directly. This might be achieved by introducing features such as buffer strips, wetlands or by modifying drainage systems.

Drainage status is known to be one of the key factors affecting NO$_3^-$ leaching and the partitioning of N losses between leaching and denitrification (Goulding, 2000). Scholefield et al. (1993) found that installing an efficient field-drainage system resulted in a three-fold increase in NO$_3^-$ leaching when compared to plots without artificial drainage. Improved drainage not only resulted in greater throughflow of water but also improved soil aeration and consequently enhanced both mineralization and nitrification during the grazing season and possibly also over autumn and winter (Scholefield et al., 1993). It has therefore been suggested that a reduction in artificial drainage or approaches to retard flow might be a way of reducing N leaching. However, reduced aeration in the rooting zone could also reduce crop yield unless managed carefully (Dinnes et al., 2002). Reducing aeration can also promote denitrification and so reducing drainage effectiveness may simply result in pollution swapping.

Vegetated buffer strips alongside water courses or at the downslope edge of fields are often seen as part of a holistic approach to improving environmental quality of runoff waters from agricultural systems as they can simultaneously reduce the quantity of multiple pollutants (Goulding, 2000; Zhang et al., 2010). Buffer strips allow N carried in throughflow (above and below ground) to be transformed by various processes including plant uptake, microbial immobilization, soil storage, groundwater mixing and denitrification (Lowrance et al., 1997; Korom 1992; quoted by Mayer et al., 2007). In the case of runoff, control of erosion and the filtering of particulate forms of N are the main mechanisms with which N loss is reduced (Mayer et al., 2007).

Natural or constructed wetlands can be used as biological treatment systems for NO$_3^-$ removal (Ingersoll & Baker, 1998; Vymazal, 2007). Constructed wetlands utilize the processes that occur in natural wetlands through the use of wetland vegetation, soils and their associated microbial communities (Vymazal, 2007). Both N uptake and the promotion of denitrification reduce water nitrate concentrations and hence if denitrification is partial then they will simply lead to pollution swapping of NO$_3^-$ for gaseous N$_2$O.
Effectiveness

Numerous studies have indicated that raising a water table into the upper soil profile layers by modification of drainage systems (known as controlled drainage) reduces the NO$_3$ concentration in ground water (Dinnes et al., 2002). Controlled drainage works by a combination of these mechanisms:

1. Increasing the anaerobic volume of the soil profile, thereby increasing N losses by denitrification
2. Decreasing the volume of drainage water exiting the drainage system
3. Decreasing the depth of the soil profile through which water infiltrates to reduce leaching potential of soil NO$_3$.

In one study, the application of a controlled drainage treatment with a water table maintained at 0.4 m throughout the growing season by sub irrigation increased crop N uptake (on average 13% for maize and 62% for soybean) compared to a subsurface drainage treatment with no drainage control (Fisher et al., 1999; cited by Dinnes et al., 2002). The mean soil NO$_3$ concentration at the 30-75 cm depth over a two year period was reduced by 46% relative to the subsurface drainage treatment with no drainage control (Fisher et al., 1999; cited by Dinnes et al., 2002). Drury et al. (Drury et al., 2009) also reported the benefits of controlled tile drainage (CD) and controlled tile drainage with subsurface irrigation (CDS) for mitigating off-field nitrate losses and enhancing crop yields. At a rate of 150 kg N ha$^{-1}$ on a maize crop followed by 0 kg N ha$^{-1}$ on soybean, CD and CDS reduced average annual N losses via tile drainage by 44 and 66% respectively relative to unrestricted tile drainage. At a higher rate of 200 kg N ha$^{-1}$ applied to maize and 50 kg N ha$^{-1}$ applied to soybean, the average annual decreases in N loss were 31 and 68%, respectively.

Two meta-analyses have studied the effectiveness of buffers strips in reducing N loading from waste waters. Mayer et al. (2007) studied both surface and subsurface water flow whilst Zhang et al. (2010) focused on runoff. In the meta-analysis of Mayer et al. (2007), the buffers, ranging in width from 1 to 220 m, were effective at removing large proportions of N contained in the water flowing through these zones; the mean removal percentage was 67.5% ± 4.0 ($N = 88$; Mayer et al., 2007). In the meta-analysis of Zhang et al. (2010), buffer widths ranged from 0.5 to 35 m, although more than 96% of the buffers studied for N removal were < 20 m wide. As with Mayer et al. (2007), N removal efficiency had a wide range: and the median N removal efficiency was 68.3%.

Increasing buffer width increases the effectiveness of the buffer in reducing N concentrations from run-off (Zhang et al., 2010) so that buffer width explained about 44% of the variation in N removal efficacy. However, buffer width is less important in reducing N from sub-surface flow. In the meta-analysis of Mayer (2007), the majority of narrow buffers (< 15 m) were effective in reducing N flow to water bodies, six of the 33 studied by Mayer et al. (2007) actually added N. It was thought that these cases were likely to be short-term events caused by nitrification or high rainfall that lead to rapid N inputs (Dillaha et al., 1988; Magette et al., 1989; Sabater et al., 2003 quoted by Mayer et al. 2007). Vegetation type and depth of rooting had very little impact on the effectiveness of the buffer strips; however having active growing vegetation present was important for effectiveness (Mayer et al. N-TOOLBOX D1.1 [41])
It is notable that while buffer strips are usually effective when new, their efficacy may decrease as they age (Wenger 1999; quoted by Zhang et al., 2010).

In the field, wetlands have been shown to be relatively efficient at removing nitrate with up to 47% efficiency (Vymazal, 2007). In a laboratory wetland microcosm experiment, NO$_3$ removal efficiencies were found to range from 8% to >95%, depending on environmental factors (Ingersoll & Baker, 1998). Removal efficiencies decreased with increasing inflow of water (ranging 5-20 cm day$^{-1}$) and increased with increasing C addition rates (ranging 1-6 g wk$^{-1}$ dried plant residue; Ingersoll and Baker 1998). Ingersoll and Baker (1998) suggested that the efficiency of NO$_3$ removal could be increased if plant growth in the wetland was increased and/or if the plants were cut and their residue left in place to provide a microbial energy source.

**Factors affecting implementation**

There are some substantial limitations to the application of drainage control (Dinnes et al., 2002). In general, it can only be applied to landscapes with a 1% slope or less due to the costs of these structures (Evans et al., 1992; Shirzadi et al., 1992; Skaggs and Chescheir 1999; cited by Dinnes et al., 2002). In addition to structural costs, drainage control on landscapes with steeper slopes would require increased management, making this strategy impractical to implement (Skaggs and Chescheir 1999; cited by Dinnes et al., 2002).

Zhang et al. (2010) reported that the soil drainage type (well drained, moderately drained and poorly drained) did not significantly affect N removal efficacy. Increased infiltration could however allow pollutants to reach groundwater in areas with highly permeable soil and shallow groundwater tables (Zhang et al., 2010).

Buffers should be protected against soil compaction that might inhibit infiltration or disrupt water flow patterns (Dillaha, et al., 1989; NRC, 2002 quoted by Mayer et al. 2007).

In designing modified drainage schemes or implementing buffer strips it is important to determine the main transport routes for nitrate through the soil profile. Changing drainage may simply redirect excess rainfall from throughflow to surface runoff or vice versa, which may have important consequences for N loss pathways, but may not result in overall lower N losses.

**Further research needed**

There was insufficient data available to study the effects of buffer slope on N removal from runoff (Zhang et al., 2010). Studying the effects of slope would aid in obtaining a more thorough understanding of N removal by vegetated buffers (Zhang et al., 2010).

There have been few full N balance studies of engineered wetland ecosystems and hence no full cost-benefit analysis can be carried out. If pollution swapping is to be avoided, denitrification should be complete (i.e. to N$_2$ rather than N$_2$O).
It is important that any end of pipe approaches do not simply result in pollution swapping of nitrate for N\textsubscript{2}O. The effectiveness of such approaches with regard to all N loss pathways needs to be considered in any research studies.
The strategies to reduce N losses to water from agriculture summarized up to this point have mostly been studied in isolation i.e. as a single strategy. This is necessary for screening of strategies to determine which will be most effective. However, we acknowledge that there will rarely be one single strategy that adequately addresses the problem of N losses to water on a given farm. Ideally, a range of strategies, or a “network of partial solutions” will be adopted that significantly reduces N losses to water. In many cases this network of solutions will also address a range of other management goals including:

a. Increased efficiency of purchased input use, particularly N fertilizer, and improved profitability

b. Improvements in soil quality through better organic matter management (crop rotations, tillage) and subsequent reductions in energy use for tillage

c. Reduced emissions of GHGs both directly, through improved efficiency of N use on the farm and reduced N\textsubscript{2}O emissions, and indirectly through a reduction in reliance on off-farm inputs (e.g. substitution of legumes as an N source for purchased N fertilizer)

d. Improvements to animal health due to better pasture management, cleaner drinking water (e.g. when waterways are fenced and alternative watering systems installed), and improved, drier footing when feeders and watering troughs are moved more frequently

Some studies have been conducted to look at the additive effects of whole farm integrated solutions to minimize N losses from agriculture. Jayasundara et al., (2007) studied the fate of fertilizer \textsuperscript{15}N applied to corn and winter wheat under conventional management practices compared with BMPs in a corn-soybean-winter wheat rotation. The experiment was conducted at the Elora Research Station, Ontario, Canada in an imperfectly drained silt loam soil from May 2000 to April 2005. In the best management system the following combinations of practices were implemented:

- No tillage;
- Side-dress application by injection of liquid fertilizer based on a soil NO\textsubscript{3}-N test and consideration of N credits from soybean when applying N to succeeding crop; and
- the under-seeding of cover crops (red clover – Trifolium pretense L.) when possible.

The paper reports that:

Over the entire period (4 years) the total NO\textsubscript{3}-N leaching loss in the conventional system was 133 kg N ha\textsuperscript{-1}, of which ca. 90% (119 kg N ha\textsuperscript{-1}) originated from soil derived mineral N. In contrast, total NO\textsubscript{3}-N leaching loss in the BM system was 68 kg N ha\textsuperscript{-1} for the entire period, about 50% lower compared to that in the conventional system. All of this was soil derived mineral N, as fertilizer derived N consisted of only 2% of the leaching loss in the BMP system.
Lobell (2007) has looked at the uncertainties associated with soil N supply and crop N demand and the impacts that these uncertainties have on fertilizer N rates. They showed that:

Eliminating uncertainty in soil N supply (but not crop demand) would reduce average N rates by 5-15% in typical irrigated rice systems, 10-30% in wheat, and 20-40% in maize. Perfect knowledge of potential crop N demand (but not soil supply) would reduce rates by 3-10% in all systems. Simultaneous knowledge of both factors reduced N rates by significantly more than the sum of their individual effects, reflecting important interactions between supply and demand uncertainties...

Furthermore, Lobell concludes that if these uncertainties could be addressed with site-specific N management, this “could lead to substantial reductions of N rates without yield loss in a wide range of cropping systems, thereby improving profitability and environmental quality”.

This is just one example of the dramatic improvements in fertilizer N use efficiency and reductions in N losses to water that can be achieved when a sensible combination of solutions is applied to a farming system. Further studies are needed that include the whole N balance i.e. take into account gaseous losses of N and the build up of pools of soil organic N, and that also include life cycle analyses of different environmental measures (in particular CO₂ emissions). Economic cost-benefit analyses are also necessary to determine the impacts on profitability of different strategies. These studies will provide valuable additional information that can be used to prioritize on-farm solutions to the problem of N losses to water from agriculture.

5. Conclusions

This literature review has provided an initial documentation of the status of knowledge on a range of categories of strategies to reduce N losses to water from agriculture. The review was designed primarily to collate the range of strategies available and to begin to extract the numeric data related to the actual impacts of each strategy on N losses to water. In future workpackages specific categories of strategies will be selected for more in-depth meta-analysis and evaluation of efficacy. This information will be used to help describe the pros and cons of each category and in some cases to prioritize the strategies within a given category.
6. References


*N-TOOLBOX D1.1*


