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Can a farming systems approach help minimize nitrogen losses to the environment?

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Abstract

Farming practices are often developed to tackle one aspect of a problem without due regard to other possible consequences. Nutrient management systems commonly concentrate on maximizing yields rather than attempting to balance the needs of the crop with the desire to minimize contamination of the environment. Particular problems are shown to be associated with utilization of animal manures on arable crops, and the move towards soil conservation practices, especially the use of cover crops. We suggest that nutrient budgets provide a suitable format for developing a systems approach. We consider the issues associated with managing nitrogen for whole farm systems, taking account of the need for efficient production and protection of the environment from losses by leaching and in gaseous form. Finally, we suggest a role for expert groups in developing a systems approach.

1. Introduction

The application of nitrogen fertilizer is cost-effective for many crops because the increase in the value of the saleable yield readily exceeds the input costs. Addiscott et al. (1991) showed that the ratio of gross profit to input for wheat was between 5:1 and 6:1. At current prices, the comparable ratio for corn in Ontario is about 3:1. Considerable research is still conducted to determine the effects of N on the growth and yield of crops. For the years 1965 and 1966, ~ 18% of reports published in *Agronomy Journal* related to N fertility, and 90% of those were concerned with effects on growth and yield. A similar proportion of papers published in 1992 and 1993 related to N fertility, and ~ 63% of these considered effects primarily on plant growth or yield.

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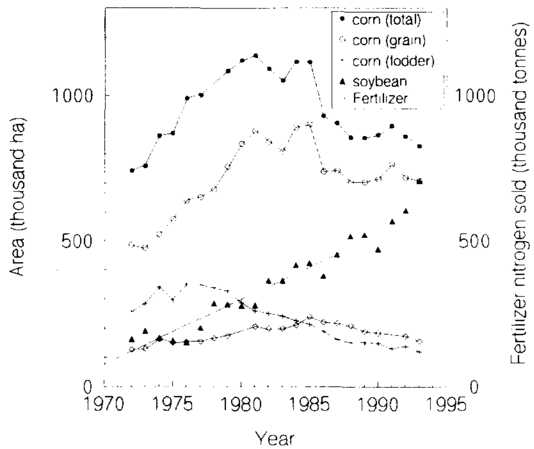

Fig. 1. Area under corn and soybean, and fertilizer sales in Ontario, 1972–1993.

The sales of N fertilizer in Ontario have steadily declined from the peak values in the mid-1980's, mainly because of a reduction in the area under corn, particularly silage corn, and an increase in soybean production (Fig. 1). Another factor has been the greater adoption of crop rotations (Coleman and Roberts, 1987), where heavier yields can be maintained for a given N application (Crookston et al., 1991). It is commonly perceived that increased reliance on industrial fertilizers has been the main cause of increased nitrate contamination of water resources (Addiscott et al., 1991). Based on this argument, a reduction in sales of N fertilizer is important for improving water quality. However, over the last 10 years, other management practices have been encouraged which can alter the consequences of reduced sales. For example, there has been an increase in the area of the Province where conservation tillage practices are employed. This latter development was stimulated by a Federal–Provincial initiative for soil conservation, the Soil and Water Environmental Enhancement Program, which aimed to reduce the phosphorus loading to the Great Lakes. Conservation tillage, particularly no-till, can result in greater losses of fertilizer N by leaching following spring rains (Gilliam and Hoyt, 1987; Goss et al., 1993a; Shipitalo and Edwards, 1993). Application of animal manures is also more problematic in conservation tillage systems because incorporation is more difficult, and volatilization of ammonia can result in the loss of much of the mineral N applied where slurry is left on the soil surface (Bless et al., 1991).

These observations support the view that agricultural research on N, and policies for implementation of improved practices, most commonly address single issues, and do not consider the consequences for the whole farming system. Similar concerns have been expressed about policy conflicts in the U.S.A. (Marks and Ward, 1993).

This paper considers the framework for the management of N over the whole farm enterprise to meet the dual aims of efficient agricultural production and protection of the environment. In considering this framework we highlight the gaps in knowledge that

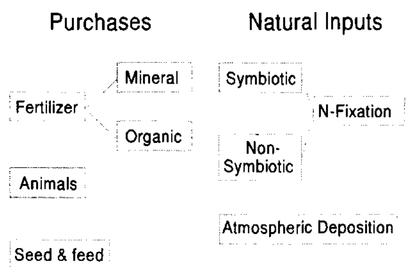

Fig. 2. Nitrogen inputs to farms.

limit our ability to predict the impact of changes to the system. Finally, we suggest a role for expert groups in developing a systems approach.

2. A framework for N management systems

A comprehensive N management system needs to consider the N cycle as it pertains to the local land unit. Information on the magnitude of the fluxes of N within the cycle need to be established. A budgetary approach appears to offer a convenient format (Schepers and Fox, 1989; Bacon et al., 1990), and is the basis of many computer models available for developing nutrient management schemes at the level of whole farms or regions. For agricultural systems, inputs are often particularly significant and are derived from numerous sources (Fig. 2).

Mineral fertilizers form the largest input to many farms in Ontario. Here as in many other regions, the burning of fossil fuels has increased the load of N compounds in the atmosphere, and ~ 20 kg ha^{-1} N returns annually through wet and dry deposition to form an input to farms (Barry et al., 1993). Another major input is the fixation of atmospheric N_2 gas by grain legumes such as soybeans [*Glycine max* (L.) Merr.], and forage legumes such as alfalfa (*Medicago sativa* L.). Purchased feed may be the largest source of N input on many livestock farms (Bacon et al., 1990).

The balance of N on the farm depends on the outputs, including losses to the environment. The outputs are summarized in Fig. 3. Major outputs are in the animal and plant protein sold off the farm. Losses to the environment, either in gaseous form by ammonia volatilization and denitrification or through leaching, are likely to exhibit more

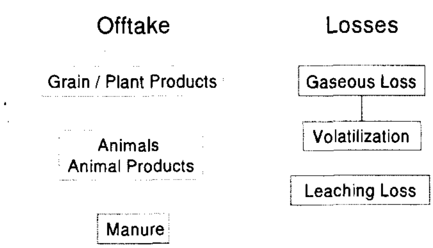

Fig. 3. Major nitrogen outputs from farms.

Table 1
The range of values for major fluxes in the nitrogen cycle with non-leguminous arable crops grown in Ontario

Process	Flux ($kg\ N\ ha^{-1}$)
Inputs:	
Fertilization	75–250
Atmospheric deposition	10–35
Outputs:	
Harvested yield	75–200
Leaching	0–75
Denitrification	0–60
Sediment in runoff	< 2
Internal:	
Mineralization	60–120
Immobilization	20–100

Values derived from Wall et al. (1982) and Barry et al. (1993).

variability than other fluxes because the conditions conducive to loss are often transitory. Losses of N in sediment and runoff are also likely to vary greatly for the same reason. Based on a study of 11 watersheds, all < 6000 ha, Wall et al. (1982) estimated that the suspended sediment leaving agricultural land in Ontario was ~ 210 kg ha^{-1} yr^{-1}. Actual values for the 11 watersheds ranged from 36 to 775 kg ha^{-1} yr^{-1}.

Assuming an organic carbon content for topsoil of 0.02 kg kg^{-1}, and a C/N ratio of 8, the average annual N loss would be only 0.5 kg ha^{-1} N, and would not be greater than 2 kg ha^{-1} N.

There are other important fluxes of N that take place mainly within the soil. The two major fluxes are associated with the processes of immobilization and mineralization. Immobilization can include the uptake of N into crop plants and incorporation of the nutrient into the soil organic matter fraction through the action of the microbial biomass. Mineralization also involves the microbial biomass, so that both these fluxes are subject to considerable variation since the biomass is sensitive to physical and chemical conditions in the rooting zone. For non-leguminous arable crops, the range of values for the major fluxes is given in Table 1.

For an agricultural system that is in a dynamic equilibrium, the fluxes associated with mineralization and immobilization are assumed to be in balance. This does not mean that in any one year the two will be equal, but over the course of a rotation there will be on average no net increase or decrease in soil N. For such systems a simplified N balance can be constructed (Fried et al., 1976; Tanji et al., 1977; Lund, 1982; Macduff and White, 1984; Hill, 1986). Any excess of N inputs over output of N in agricultural produce then represents a potential loss to the environment through leaching or in gaseous form. Barry et al. (1993) used this approach to estimate the maximum likely concentration of nitrate-N leaching from different cropping systems in Ontario. Good agreement was obtained between the predicted (7 mg L^{-1} N) and measured (10 mg L^{-1} N) concentration of nitrate-N in water draining from a cash crop farm growing corn (*Zea mays* L.), soybean and wheat (*Triticum aestivum* L.) in rotation. The importance of considering the whole farming system of a cash–crop farm was evident from studies of N leaching where concentrations of N in drainage water depended as much on cropping sequence as on the current crop (Goss et al., 1993a).

For mixed farms large annual excesses of inputs over outputs of N have been predicted using a nitrogen balance approach (Barry et al., 1993). An excess of 93.5 kg ha^{-1} yr^{-1} N was predicted for a dairy farm where no mineral N was used as fertilizer, but $455 \cdot 10^3$ L of hog manure were bought in, and cover crops were grown to help improve soil structure. The rotation used was much longer than on the cash–crop farm, consisting of spelt, followed by oilseed radish cover crop, oats undersown with red clover, barley, winter rye, followed by oilseed radish cover crop, two years of alfalfa and grass hay, and one year of alfalfa and grass pasture. Large excesses of N have also been reported for dairy farms in Europe (Aarts et al., 1992; Kaffka and Koepf, 1989).

Goss and Goorahoo (1996) investigated the N balance of seven farms: four growing cash crops, two dairy farms and one swine rearing farm. When the predicted value for the nitrate concentration in the groundwater was regressed on the measured values, results indicated an average over prediction of 33% for measured values in the range 0–28 mg L^{-1} N.

3. Current issues in N management systems

The study of farming systems (e.g., Aarts et al., 1992) has already indicated that the impacts of a number of components need to be considered in more detail.

3.1. Cover crops

There has been much interest in growing N-fixing legumes, such as red clover (*Trifolium pratense* L.) and hairy vetch (*Vicia villosa* L.), as cover crops, particularly on erodible soils. As N losses from soil erosion are relatively small (Table 1), N_2 fixation by the leguminous cover crop can have a marked impact on the potential for N leaching. Tomato/wheat appears to be a promising rotation for Ontario's field vegetable growers. The simplified N budget indicated only a small annual excess of N in the rotation (Table 2). When red clover was grown as a cover crop following the harvest of wheat, the shape of the N response curve did not change. Consequently, the optimum N fertilizer application remained the same although there was a yield benefit from the adoption of the clover. The simplified budget suggested that the additional N input to the system from the clover could have resulted in a four-fold increase in the nitrate concentration in drainage water (Table 3). One role for cover crops is to immobilize N at risk of leaching in the fall and early spring. Thus a key aspect is the extent to which leguminous cover

Table 2
Nitrogen budget for a tomato (*Lycopersicon lycopersicum* L.) and wheat rotation

Input	(kg N ha^{-1})	Output	(kg N ha^{-1})
Seed	1.4	grain/fruit	77.4
Fertilizers	62.5		
Fixation (non-symbiotic)	5.0		
Atmospheric deposition	18.4		
Total	87.3		77.4
Imbalance [a]	9.9		
Predicted groundwater contamination (mg L^{-1})	6.2		

Agronomic information taken from Johnston (1992).
[a] Total inputs – outputs, leaching loss not included.

crops exploit soil-N rather than fix atmospheric N_2. The maximum depth from which cover crops will extract significant amounts of inorganic N also needs examination.

Uptake of N by non-leguminous cover crops can reduce the mineral-N content of the soil significantly in the fall compared with leaving the soil bare. Cover crops also extract water from the soil, reducing the potential for percolation and consequent leaching compared with uncropped land. The magnitude of the effect depends on climatic conditions (Chapot et al., 1990).

The early planting of a cover crop is usually required to ensure substantial uptake of N (Meisinger et al., 1991). In Europe, planting before 1 September appears to be essential if a significant uptake of N is to take place (Landman, 1990).

In a study on the use of cover crops following the fall application of cattle manure, Goss et al. (1993b) found a good correlation between the amount of N immobilized in

Table 3
Nitrogen budget for a tomato/wheat rotation with red clover used as a cover crop after wheat

Input	(kg N ha^{-1})	Output	(kg N ha^{-1})
Seed	1.4	Grain/fruit	81.4
Fertilizers	62.5		
Fixation			
symbiotic	34.8		
non-symbiotic	5.0		
Atmospheric deposition	18.4		
Total	122.1		81.4
Imbalance [a]	40.7		
Predicted groundwater contamination (mg L^{-1})	25		

Agronomic information taken from Johnston (1992).
[a] Total inputs – outputs, leaching loss not included.

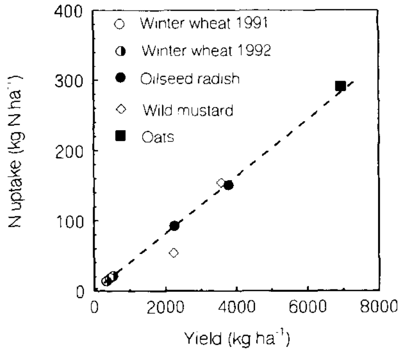

Fig. 4. Uptake of nitrogen by winter wheat, spring oats, wild mustard, and oilseed radish cover crops grown in southeastern Ontario.

the plant (y kg ha^{-1} N) and the dry matter (x kg ha^{-1}) produced in the fall (Fig. 4) given by:

$$y = 0.04x \quad (r^2 = 0.989,\ p < 0.001)$$

Remineralization of N from their residues is an important aspect in the choice of cover crops for reducing N leaching. There is some information on the total amount of N released from cover crops. The effect that growth stage of the cover crop, at the time it is killed, has on the total released has also been investigated for crimson clover (*Trifolium incarnatum* L.) (Ranells and Wagger, 1992). The timing of the release of N from cover crops is less well documented. Studies with red clover and alfalfa indicated that soil mineral N increased rapidly in spring, and in summer and fall there was a further steady increase after the phase of maximum uptake under barley and corn (Alder, 1988). Using ^{15}N as a tracer, the same author showed that most of the available N from the legume crops was taken up by barley in the first five weeks after seeding, but corn incorporated most of the N it derived from the legumes between five and twelve weeks after planting. Maitland and Christie (1989) compared the contribution to the N in barley (*Hordeum vulgare* L.) and corn from the plough-down of four legumes, hairy vetch, sweet clover (*Melilotus alba* L.), alfalfa and red clover (two varieties — Arlington® and Florex®). Corn recovered more of the N from hairy vetch, alfalfa and red clover (cv. Florex®) than did barley, suggesting that the N from these crops became available later than that from sweet clover and red clover (cv. Arlington®). Miller et al. (1992) found that N from oilseed radish (*Raphanus sativus* L. var. *oleifera*), which was killed by frost late the previous fall, was released early in the spring. Only 35% of N in the oilseed radish appeared to be recovered in the following corn crop compared with a recovery of 70% from red clover, suggesting that there was a significant loss of N by leaching or denitrification following oilseed radish, or that more of the N in the plants was not readily mineralizable. In contrast, less N was recovered by corn following ryegrass (*Lolium multiflorum* L.) than by corn after a winter fallow, suggesting that little N was remineralized from residues of this cover crop during the spring or summer.

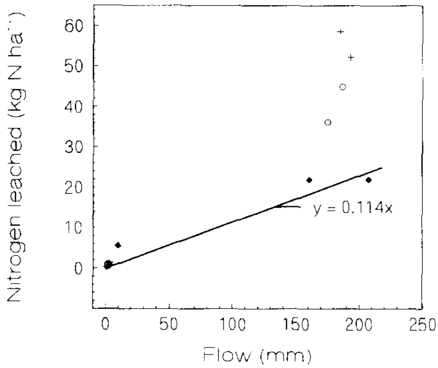

Fig. 5. Nitrate-N leached from land under fall-sown crops after spring crops preceded by winter fallow or cover crops. + = cover crop; ○ = winter fallow; ♦ = continuous winter cereals. The regression line indicates the relationship between drainage and N leached under continuous fall-sown cereals over 4 years (M.J. Goss and K.R. Howse, pers. commun., June 1993). Adverse effects of the cover crops were significant at drainage rates close to the long-term average at the site.

Mustard (*Sinapis alba* L.) and fodder rape (*Brassica napus* L.) released some N to following spring crops (Catt et al., 1992). However, results from the same experiment showed that nitrate leaching under a fall-sown cereal was greater where the preceding crop was spring-sown rather than fall-sown. Furthermore, leaching (both mass and flow-weighted mean concentration) was greater where cover crops had been grown, than where the land had been fallowed prior to sowing the spring crops (Fig. 5). More studies on the fate of N taken up by cover crops are needed before better cropping systems can be developed that reduce nitrate leaching, and prevent soil erosion.

3.2. Atmospheric deposition

Much of the variation found in the input of N from atmospheric deposition was apparently due to ammoniacal forms, and could be particularly large close to animal units (Sanderson, 1977). If the atmospheric deposition in Ontario (~ 20 kg ha^{-1} N) was omitted as an input to an arable farm where through-drainage is 200 mm yr^{-1}, the concentration of NO_3^--N in the drainage water might be underestimated by 10 mg L^{-1} N. Significant contamination of a water resource could therefore result if nutrient management was based on this incomplete assessment of inputs. Losses of ammonia by volatilization from manure systems, including application systems has been found to be very variable, and dependent on the design of the storage as well as on atmospheric conditions and soil physical conditions. On farms with animal enterprises, considerable uncertainty would be expected in the potential for nitrate leaching to groundwater because of this variability. The improved knowledge on animal nutrition has resulted in a considerable potential to reduce the amount of N excreted, and hence the ammonia at risk to volatilization. This reduction in the N content of manure can derive from the use of dietary supplements and improved formulation of feed coupled with phase feeding, which links diet to growth stage (see Goss et al., 1993c).

3.3. Utilization of animal manures

There is a major limitation to the adoption of integrated N management on livestock farms because of our inability to make adequate predictions of the nutrient availability from manure. The main limitation probably results from the variability in the gaseous loss of N, mainly as ammonia, at the different stages of manure management from excretion to the time that the manure is incorporated into the soil (Burton and Beauchamp, 1986). Losses of N during field application also vary widely according to the method used (Thompson et al., 1987).

Treatment of manures in storage or immediately prior to application, such as by acidification, solid separation, or dilution, can be effective in minimizing N losses. The use of acidification and solid separation of manures can also improve the balance of N to phosphorus and potassium in the material, making it closer to that needed by the crop (Stevens et al., 1992a). There is also evidence that these treatments of manure can enhance the efficiency of N utilization by crops (Stevens et al., 1992b). However, these approaches have yet to be shown to be practical for most farmers. Even if there is more technical development in manure treatment, the amount and timing of the availability of N released during the mineralization of the organic-N fraction of the manure is poorly understood.

3.4. Agronomy

Methods of applying N to crops, as mineral or organic fertilizers, are also important in determining the potential for N loss by leaching. Nitrate ions at the soil surface are more readily transported in mass flow towards macropores where they can rapidly leach downwards out of the rooting zone. In the spring when soils are still moist, a single surface application can leave more N vulnerable to leaching than if the application is split. Also, nitrate resulting from nitrification within structural units of the soil can be less vulnerable than nitrate applied as fertilizer (Goss et al., 1993a).

The amount of N removed in saleable produce is important in determining the potential for N leaching. Agronomic, climatic and soil factors which lead to impaired growth also tend to reduce utilization of N. Equally it is important to ensure that there is little fertilizer-N residual in the soil as crops start to mature. The combined use of soil test and fertilizer response data has successfully provided the means of achieving the latter goal for corn, although further work is necessary if organic manures (animal manure, green crops and cover crop) are used (Kachanoski and Beauchamp, 1991; Meisinger et al., 1992).

In grazed grassland the development of patchiness in the sward due to selective feeding and defecation has long been recognized. It is evident that when these grasslands are used for arable production, the distribution of nutrients in the soil will reflect the patterns of deposition and utilization established under the previous management. Heterogeneity may exist in many arable fields because of the processes of soil formation (Bouma and Finke, 1993), because of the trend to combine fields for more efficient use of machinery (Voorhees et al., 1993), and because of management practices, such as drainage and tillage. There is an increasing recognition of the need for location-specific fertilizer application in recognition of the variability in yield within fields. The availability of portable global positioning systems and the technology for variable rate delivery now makes this a practical option (Ascheman, 1993).

Soil variability can also have important consequences for nitrate leaching from fields given a uniform fertilizer application. van Nordwijk and Wadman (1992) developed a model to identify the consequences of soil variability on the amount of residual mineral N in the soil at harvest. The results suggested that spatial variability in soil properties would result in an increase in the difference between the N fertilizer required for maximum economic yield, and that which would not result in groundwater being contaminated with nitrate above the maximum acceptable for potable water. The greater the variability, the greater the amount of residual N in the soil at harvest if nutrients were applied uniformly. To minimize the effects of spatial variability in the N supply from the soil, the distribution of fertilizer N applied would have to be negatively correlated with the soil supply (van Nordwijk and Wadman, 1992).

Soil variability and variability in annual rainfall can affect the loss of N through leaching. At the same catchment the leaching loss under a given cropping system can be linearly related to annual drainage (Kolenbrander, 1981) (see also Fig. 5). Leaching loss of N after slurry application was predicted to be greater if no account was taken of soil variability than where the application was made on a soil-specific basis (Finke, 1992). Based on the evidence of reduced losses of mineralized nitrate under no-till, if soil variability results in greater by-pass flow (Kung, 1990), leaching losses could be reduced for similar values of residual mineral N.

An engineering approach to reducing nitrate leaching in soils with subsurface drainage, has been to raise the water table in summer. This has aimed to increase crop growth by eliminating soil water deficits, and reduce the risk of residual fertilizer at the end of the season. One consequence of this approach that requires proper investigation is the risk of enhanced denitrification which, through the impact of gaseous oxides of N on global warming, could have a greater environmental impact than the increase in nitrate concentrations in water. The loss of nitrous oxide during nitrification also needs better quantification for this reason (Bouwman, 1990; Eichner, 1990).

3.5. Economics

These biophysical aspects of N management have also to be considered in relation to the economics of production. The area of most need is the development of appropriate economic models that predict off-farm costs associated with the environmental impacts of N management.

4. A role for expert panels in the systems approach

The greater knowledge base now available, and the rapid development of mechanistic models dealing with the components of N management systems, requires that some effort be put towards improving our ability to integrate these components. As farmers are being put under increasing pressure from the public to reduce environmental impacts, there is considerable need to provide extension information now. In Ontario there is increasing demand for on-farm studies to supply this need. At the University of Guelph the move towards a systems approach has been aided by the establishment of expert panels. These provide a forum where researchers from different disciplines, extension personnel, commodity groups, producers and environmentalists can meet to review relevant issues. The intent is for the panels to provide critiques of current practices, assess the benefits and limitations of new ones, and to stimulate the adoption of promising research through extension and education. By these means it is hoped to revitalize the cycle of research, extension, application and identification of limitations on the system.

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