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The importance of potassium in soils and plants
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The Importance of Potassium in Soils and Plants

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Abstract

Potassium is an essential nutrient and is required in large amounts by plants, animals and humans. Consequently soil must contain adequate amounts of plant-available potassium if crops are to achieve economically viable yields of acceptable quality for human and animal diets. Potassium has two roles in plants, (i) it has an irreplaceable role in the production of proteins and sugars; (ii) it is the "plant preferred" ion (element) for maintaining the salt concentration in the water in individual cells. A high concentration of potassium maintains the water content of the cells and keeps them turgid (swollen, rigid) so that the rigid structure of the leaves is maintained. Maintaining the turgidity of leaves is important to keep the largest possible surface area exposed to sunlight to capture the energy from the sun to optimise photosynthesis (the conversion of carbon dioxide to sugars) and hence crop yield. The interrelationship between nitrogen and potassium in plant nutrition is discussed to show why nitrogen is only used efficiently when soils contain sufficient potassium. The balance between the availability of nitrogen and potassium controls not only yield but also the quality of crops used for food and feed. The mechanisms by which potassium is held in soil are discussed in relation to the fraction of soil potassium that is most available to crops.

Role of potassium in plant nutrition

As a major constituent in all living cells, potassium is an essential nutrient and is required in large amounts by plants, animals and humans. To be healthy and grow normally, animals and humans require an adequate supply of potassium from their food and feed. The majority of this potassium comes directly from plants. Thus, not only to meet the needs of the plant but also that of animals and humans, it is essential that an adequate supply of potassium is available in the soil to optimise the yields of crops and grass containing suitable amounts of potassium. Although there are abundant amounts of potassium in the rocks comprising the surface of the earth, it is potassium in the clay fraction of soils, together with that in soil organic matter, which is mainly available for uptake by plant roots. As the clay particles disintegrate (weather) potassium is released. The amount and rate of release of the potassium from clay depends on the nature of the rock from which it came. However, even with clays which weather quickly, the amount of potassium released is invariably too little to meet the potassium requirement of grass and crops with a large yield potential. Thus it is essential to supplement soil potassium supplies to ensure that crops produce economically viable yields of acceptable quality and at the same time maintain soil fertility to benefit future production.

In plants, animals and humans, potassium always occurs as a positive ion (K^+) in every living cell and it is the major mineral ion within cells. Potassium is also found as the K^+ ion in soils, many rocks, surface and seawater. Potassium ions are always balanced by an equal number of negative ions. For example, solid potassium chloride, (muriate of potash, KCl), when dissolved in water dissociates (breaks apart) into equal numbers of K^+ and Cl^- ions (Cl^- is a negative ion). In this paper, when discussing the role of the element potassium in crop nutrition and soil fertility, potassium will be shortened to K. Where reference is made to potassium applied as fertilizers, the term potash is used, i.e. K_2O, the oxide of potassium. For quick conversion, 1 kg K is equivalent to 1.2 kg K_2O.

In cells, K has an irreplaceable function primarily concerned with the activation of enzymes that play a key role in metabolic processes, especially the production of proteins and sugars. In this biochemical function, K cannot be replaced by any other ion, however the amount of K required is small. A much larger concentration is required to maintain the water content and hence the rigidity (turgor) of each cell – a biophysical function. In grass probably not more than 0.8% K in dry matter is required for the biochemical functions but at least 1.6% K in dry matter for the biophysical function (Barraclough and Leigh, 1993). A large concentration of salt in the cell sap (the water inside the cell) creates conditions that cause water to move into the cell (osmosis) through the porous cell wall. This passage of water between cells, depending on the salt concentration in the cells, is essential for the cells to function properly and to take up water from soil. The condition that leads to water movement is termed the “osmotic potential”. Potassium is the “plant preferred” salt for maintaining the osmotic potential of plant cells. When plants take up water by osmosis the cells gradually become more turgid (swollen, rigid) until no more water can enter them. Turgidity is very important to plants because it maintains the rigid structure of most annual crops, which do not have a woody structure of trunks and branches.

For plants maintaining the turgidity of the leaves is important to maintain the largest possible surface area exposed to sunlight to optimise the process of photosynthesis in the green chloroplasts in the cells. Photosynthesis is the process by which plants harvest energy from the sun to convert carbon dioxide to sugars. The carbon dioxide in the air has to enter the leaf through the stomata – tiny openings mainly on the underside of the leaf. These tiny openings are surrounded by “guard cells” and it is only while they are turgid that the stomata remain open and allow carbon dioxide to enter the plant. The ready access into the leaves of carbon dioxide to be converted to sugars is essential to maximise yield. However, there is something of a conflict of interest here because the plant loses water mainly through the stomata and it needs to close them when too little water is available. The plant can close the stomata by lowering the concentration of K in the guard cells, and the visual symptom is wilting of the leaves. Thus the balance between K supply and water is essential for the well-being of the plant.

A high osmotic potential in plant cells is also needed to ensure the movement through the plant of nutrients required to sustain growth and the sugars produced by photosynthesis. These sugars are transported within the plant, with the aid of K, to where they are required to increase vegetative growth, for example grass leaves, and later in the growing season to storage organs. In storage organs, like cereal grains and potato tubers, sugars are converted to starch; in sugar beet roots sugars are stored as such. In its biophysical role, K can be replaced by another cation including, sodium, calcium and magnesium but the evidence is that most plants prefer to use K. Plants can also use organic ions, containing carbon, to replace K^+. If this occurs because there is a deficiency of K in the soil, yield will be lost because carbon that should be going to increase growth will be diverted to produce organic acids to maintain turgor.

Potassium has other important roles in crop production. Very importantly it increases the efficiency of use of nitrogen (N), and this is discussed in more detail in the following section. Potassium also has a beneficial effect on the quality of crops (Mengel, 1997; Wiebel, 1997), especially more and better quality protein. There is evidence that adequate K in the plant can minimise the adverse effects of pests and diseases (Krauss, 1997) and reduce abiotic stress, including drought and cold stress (Marschner, 1995).

The interaction of nitrogen and potassium

The primary processes involved in growth and dry matter production, such as photosynthesis and protein synthesis, upon which final crop yield depends, occur within individual cells and tissues. Individual cells are extremely complex and highly specialised to perform specific functions in the development of the plant. However, they can be visualised simply as an expandable wall enclosing as central space, the vacuole, containing an aqueous solution, cell sap. The vacuole, because it is composed largely of water, has an important role in the water economy of the plant as noted above. It is also within the vacuole that nutrients like potassium, phosphorus and magnesium are stored together with sugars produced by photosynthesis. The vacuole of a mature cell comprises more than 80-90% of the total cell volume and contains most of the water in the plant. As cells expand the volume of water they contain increases considerably. For example, a 2-fold increase in cell length, approximately results in an 8-fold increase in cell volume and thus there is a large increase in the quantity of water within the cell.

One major determinant of plant growth and a prerequisite for large yields of grass and arable crops is the rapid expansion of the leaves (the leaf canopy) in spring to fully capture sunlight energy required for photosynthesis. Nitrogen is the major driver of leaf canopy expansion and applying N to any field-grown crop invariably results in a very obvious visual response, which is often associated with an increase in yield. The increase in leaf canopy expansion is achieved by an increase in cell division producing more cells and by an increase in size of individual cells. This N-induced increase in cell number and volume is accompanied by an increase in cell water content and this requires a corresponding increase in the uptake of K to maintain the osmotic concentrations of the leaf tissues at a level sufficient to maintain turgor and therefore dry matter production. Much of the total amount of N and K required by crops is therefore taken up to sustain development and expansion of the leaf canopy. This will be in the early months of growth of most arable crops but for much longer periods for grass which is being harvested for silage or hay or constantly eaten by grazing animals. The amount of K taken up by a crop depends on the input of N where the soil K supply is adequate; and most grass crops actually contain more K than N. The visual response in colour and/or growth of a crop to applied N is seen frequently and is often taken for granted. However, what is not seen visually, and is thus rarely realised by farmers and advisors, is that applying N leads to an increase in the amount of water in the crop and thus a requirement for more K. For example, the difference in water content between cereal crops well and poorly supplied with N can be between 10-15 t/ha, while for sugar beet well and poorly supplied with N, the difference is much larger 30-35 t/ha.

Soil potassium supply

Plant roots take up K from the water in the soil (the soil solution) when growing vigorously. A rapidly growing cereal crop will take up as much as 6 kg K per hectare (ha) per day and sugar beet even more, up to 8 kg/ha daily. To maintain this rate of uptake, the amount of K in the soil solution has to be replenished quickly. This is only possible if the soil contains a sufficiently large supply of readily plant-available K. Most of these reserves, which constitute an important aspect of soil fertility, have accumulated from past applications of fertilizers and manures. These reserves must be maintained by applying fertilizers and manures containing K.

When used wisely, N is the major agronomic input to achieve crop growth. But for N to play is full roll in increasing yield and justify its cost, a crop must have access to, and take up, an adequate amount of K from the soil and this K must be readily available for crop uptake. It is now generally accepted that on many soils, K (and phosphorus, P, also) can accumulate as plant-available reserves from applications of fertilizers and organic manure and from crop residues. These plant-available reserves of K are held in soil by ion exchange reactions. The positively charged K ion (K^+) is held on negatively (-) charged sites on soil constituents, usually clay and silt particles, and organic matter. Depending on the position of the negatively charged sites within the soil matrix, and hence the physical position of the K, it can exchange with positively charged ammonium ion (NH_4^+) when the soil is extracted with a solution of an ammonium salt. Thus this K is called exchangeable K (K_{ex}) and it is K often held at the edges of clay particles and on organic matter. Many, many field experiments with a wide range of crops over many years have shown that the response of a crop to applied K fertilizer correlates well with K_{ex} – response decreasing as K_{ex} increases. In addition crop yield can be related to K_{ex} and in this case yield increases as K_{ex} increases. When K is held within the clay particles and will not exchange with NH_4^+ it is known as non-exchangeable K. However, many field and laboratory experiments have shown that this non-exchangeable K can over time become available to crops.

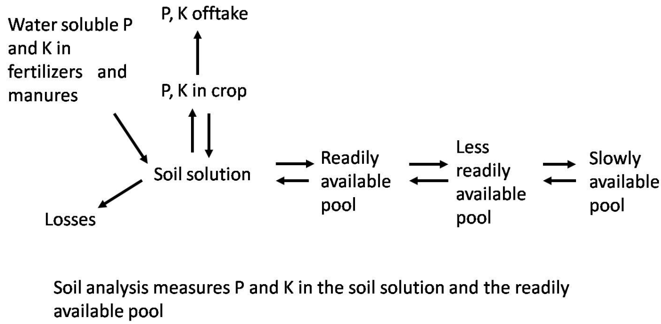

Figure 1. Current concepts about the behaviour of potassium in soil.

Conceptually K can be visualised as occurring in various pools within the soil which can be related to its availability to plants and its exchange with ammonium ions (Figure 1). Thus K in the soil solution is immediately available for uptake by plant roots, while that in the readily available pool is K which is exchangeable with NH_4^+ ions. The less readily available pool contains K that is not immediately exchangeable (non-exchangeable K). This K has mostly accumulated from past applications of fertilizers and manures. K in the fourth pool is only very slowly available and is mostly K within the mineral matrix of the soil; the release of this K depends on the weathering of the clay particles. The important feature in Figure 1 is the reversible transfer of K between the first three pools. Thus as K is taken up by roots and removed from the soil solution, it is replaced by K from the readily available pool. This exchangeable K, in turn, is replaced by K that is not immediately exchangeable and is held in the less readily available pool. The reverse process occurs when K is added to soils in fertilizers and organic manures. Any K not taken up by a crop increases the K in the readily available pool. Some of this K migrates to sites within soil constituents where it is not immediately exchangeable to ammonium ions. Such changes are well illustrated by data from an experiment at Rothamsted on a silty clay loam soil (Table 1). Clover was grown each year, and each year the amount of K applied and that removed in the crop was known so that the K balance could be calculated (K balance = K applied *minus* K removed). Soil samples were taken periodically and exchangeable K, i.e. the K in the readily available pool, was determined. In the years when the K balance was positive, the increase in the amount of exchangeable K was less than the K balance; in the years when the K balance was negative the decrease in exchangeable K did not account for all the K removed in the harvested crop. Thus when the K balance was positive, K was moving into the less readily available pool; when the K balance was negative, K was released from the non-exchangeable pool and was taken up by the crop.

Table 1. Potassium balances and changes in exchangeable K in the 0-23cm topsoil of a silty clay loam during 1956-1983. Garden Clover experiment, Rothamsted. (Adapted from Johnston et al., 2001a).

Period	K applied *annually* (kg/ha)	Total K balance *in period* (kg/ha)	Exchangeable K at start and end of each period (kg/ha)			Change in exchangeable K as % of K balance
			Start	End	Change	
1956-66	none	-246	171	194	23	-
	136	617	171	431	260	42
1968-78	250	1667	375	1065	690	41
1979-83	125	-1494	1065	502	-563	-38

Based on this concept of K being held in four pools, related to the availability of the K for uptake by plant roots, the question to be asked is how much K should there be in the readily available pool. Fortunately, the amount of K in the readily available pool can be estimated by determining the exchangeable K. This is done by a well established method of analysis in which the soil is leached or shaken with a solution of an ammonium salt, usually ammonium nitrate or ammonium acetate. Data from very many field experiments show that as the exchangeable K in soil increases, the rate at which yield increases declines until the yield reaches a maximum. The level of exchangeable K at which the yield approaches the maximum (the asymptote) can be taken as the critical value for that crop, soil type and farming system. Below the critical value, yield is lost a financial penalty for the farmer; above the critical value there is no point in adding more K because there will be no further increase in yield. The best fertilizer practice is to increase soil exchangeable K to the critical level and then maintain it at that level by replacing the K taken off in the harvested crops. To check that this approach is working, the soil should be sampled every 3-5 years to check that exchangeable K is kept near the critical value.

Estimating potassium sufficiency using crop analysis

Crop analysis has been proposed as a method of estimating the nutrient supplying capacity of a soil. This can work well for some nutrients, especially micronutrients (trace elements), where there are clear visual symptoms that correlate well with analytical data so that sufficiency/deficiency levels can be clearly identified. Visual deficiency symptoms for the major nutrients can be recognised but they can often be confused, for example N and K deficiency in young, actively growing green leaves is frequently not attributed correctly. Also for the major plant nutrients, it is not always possible, in fact it is almost impossible, to relate deficiency of one of these elements to its concentration in the dry matter of plant tissue. This is because the concentration of the major nutrients, when expressed as a percent in crop dry matter, declines rapidly throughout growth. Thus levels of sufficiency and deficiency cannot be compared unless they can be related to very specific growth stages.

However for K, because it has an important osmotic role in plants (Wyn Jones, Brady and Spiers, 1979), it has been suggested that its concentration should be expressed on the basis of its concentration in tissue water, not as a percentage in dry matter (Ahmad and Wyn Jones, 1982). Leigh and Johnston (1983a, 1983b) related changes in K concentrations in tissue water of spring barley grown in the field and grain yields at harvest and studied the effects of varying levels of exchangeable K, fertilizer N inputs, drought and available P status of soil on tissue water K concentrations. They found that from plant emergence until the crop began to loose water as it ripened, the K concentration in tissue water remained essentially constant and the actual concentration was affected only by the level of exchangeable K in soil and not by any of the other factors examined. Thus when sampled at any point during growth up to anthesis, expressing the K concentration in tissue water could indicate whether the soil was supplying sufficient. K.

The usefulness of expressing K concentrations on a tissue water basis for grass has been discussed by Barraclough and Leigh (1993). Over a four-year period, in a long-term grass experiment at Rothamsted they showed that average maximum dry matter yield was obtained at 114 mg/kg K_{ex}. Grass well supplied with K, i.e. grown on soil with more than 120 mg/kg K_{ex}, maintained an average K concentration of about 200 mM K in tissue water of whole shoots in spring. Yield was decreased by about 60% on soils with only 60-90 mg/kg K_{ex}, and the K concentration in tissue water was only about 64 mM. There was some variation in the K concentration in tissue water for grass grown on soils well supplied with K which was related to the availability of water. However, this method of expressing K concentrations in tissue water, rather than dry matter, could be combined with soil analysis. If there is doubt as to whether a soil with little exchangeable K is releasing too little K, a K concentration less than 150 mM in grass tissue water and 200mM in cereal tissue water would confirm that the crop was likely to suffer from K deficiency.

Response by crops to potassium

When assessing the benefit to yield of potassium it is essential to ensure that other nutrients are not limiting yield. Table 2 shows annual grass yields over six years, 1958-63 on three soils with a range of amounts of Olsen P and exchangeable K. Reserves of P and K in two of these plots had accumulated in an experiment between 1848 and 1951. Maximum yield was on the soil with most available P and K but which of the two nutrients was more important is difficult to tell. In this period K_{ex} in all three soils declined to almost the same level.

Table 2. Effect of phosphorus and potassium reserves in soil on the average annual yield of dry grass, the nutrients removed and the response to applied K, Agdell experiment, Rothamsted. (Adapted from Johnston et al., 2001b).

	Soil 1	Soil 2	Soil 3
Years 1958-63			
Olsen P (mg/kg), 1958	17	6	2
Exchangeable K (mg/kg), 1958	185	152	109
Annual yield dry grass, t/ha	6.93	6.58	3.22
P_2O_5 removed each year (kg/ha)	28.9	23.6	8.7
K_2O removed each year (kg/ha)	165	140	66
Years 1964-69			
Exchangeable K (mg/kg), 1958	185	152	109
Exchangeable K (mg/kg), 1964	111	103	97
K_2O applied (kg/ha), 1964	Annual yield dry grass (t/ha)		
0	7.50	6.59	6.35
315	8.71	8.12	8.23
630	8.22	7.96	7.98

The effect of adding K fertilizer depends on the level of K_{ex} in soil, but the amount and rate of release of non-exchangeable K can also have an effect. The three soils cropped with grass between 1958 and 1963 in Table 2 above continued in grass for the next six years, 1964-69. Now however, the plots were divided to test a range of additions of potash (Table 2). The relationship between yield and K_{ex} was better when the K_{ex} values for 1958 rather than those for 1964 were used. This suggests that non-exchangeable K built up between 1848 and 1951 was being released slowly.

Applying fresh K to soils with less than adequate K_{ex} does not always increase yields to those on soils with sufficient K_{ex} (Table 3). In the top part of Table 3, fresh K increased yield of spring barley but not of potatoes and sugar, from beet, on the poor soil to that on the better soil. In the bottom half of the Table 3, the yield of spring barley on all four soils was very similar. But yield of field beans and potatoes grown on soils deficient in K was much less than on soils with adequate K_{ex} and adding fresh K to these soils did not increase yields to those on soils well supplied with K. Crops like potatoes, sugar beet and field beans are responsive to K and yield better on soil well supplied with K. It is perhaps somewhat surprising that field beans behaved more like potatoes than cereals. For cereals, K uptake presumably accompanies nitrate uptake as a balancing cation, in beans N is fixed in the nodules and transferred directly to the root so that large amounts of K must be in the soil solution so that it is taken up as hydrogen ions are excreted.

Table 3. Effect of exchangeable K in topsoil on yield of three arable crops and response to applied K fertilizer. (Adapted from Johnston et al., 2001b)

1973-76 on a silty clay loam, Agdell, Rothamsted					
		Exchangeable K			
Cropping	K_2O applied (kg/ha)	85 mg/kg Yield (t/ha)		115 mg/kg Yield (t/ha)	
		no K	with K	no K	with K
Spring barley, grain	60	5.39	6.15	5.39	5.22
Potatoes, tubers	240	7.4	23.9	23.0	27.2
Sugar beet, sugar	305	3.64	4.75	4.83	4.96
1977-80 on a sandy clay loam, Saxmundham, Suffolk					
		Exchangeable K			
	K_2O applied (kg/ha)	113 mg/kg Yield (t/ha)		166 mg/kg Yield (t/ha)	
		no K	with K	no K	with K
Spring barley, grain	63	5.68	5.68	5.71	5.86
Field beans, grain	63	2.52	3.60	4.42	4.38
Potatoes, tubers	224	28.8	39.6	43.1	44.0

Nitrogen use efficiency and potassium supply

Today there is much talk about the efficient use of applied N fertilizers. In part this can be related to their cost to the farmer and in part to the fact that nitrate-N lost by leaching or by denitrification to nitrous oxide, a greenhouse gas, has adverse environmental effects. Table 4 shows the response by spring barley to three amounts of applied N at two levels of K_{ex}. The difference in yield increased to about 2 t/ha as the amount of applied N increased. On the soil with least K_{ex} it was not justified to apply more than 48 kg N/ha.

Table 4. Effect of exchangeable potassium on the efficiency of nitrogen use by spring barley. Hoosfield Barley, Rothamsted. (Adapted from Johnston et al., 2001a)

Grain Yield (t/ha)		N applied (kg/ha)			
		0	48	96	144
Exchangeable K (mg/kg)	60	1.8	3.1	3.1	2.9
	340	1.9	3.9	4.9	5.0

The response of potatoes and sugar beet to increasing amounts of N fertilizer on two soils well supplied with P but with different amounts of K_{ex} shows that the yields of potatoes were always larger on soil with more K_{ex} (Table 5). It was not justified to apply more than 144 kg N/ha to potatoes on soil with little K_{ex}, to apply more was to use it inefficiently. Sugar beet is a deep-rooted crop and tends to search soil for nutrients more effectively than potatoes. Consequently, the beet found enough K on the soil with less K_{ex} to meet their requirements for K when 144 kg N/ha was given but not 216 kg N/ha. When this larger amount of N was applied a larger amount of K_{ex} was needed to use the N efficiently.

Table 5. Response of potatoes and sugar beet to nitrogen on two soils: one given P only; the other P and K. 1969-73, Barnfield, Rothamsted. (Adapted from Milford and Johnston, 2007)

	N applied (kg/ha)			
Treatment per ha each year	0	72	144	216
	Potato tubers (t/ha)			
76 kg P_2O_5	9.9	17.0	24.0	25.4
76 kg P_2O_5 + 270 kg K_2O	11.0	21.2	28.4	35.7
	Sugar beet, sugar (t/ha)			
76 kg P_2O_5	13.5	24.6	36.2	37.6
76 kg P_2O_5 + 270 kg K_2O	14.6	25.3	37.0	44.6

A very similar benefit from having enough K_{ex} in soil was found for winter wheat (Table 6). Yield was always larger on the soil with more K_{ex}, and more N was recovered in the grain.

Table 6. Effect of soil potassium and applied nitrogen on the yield and nitrogen content of winter wheat grown on a sandy clay loam. 1983-84, Saxmundham. (Adapted from Milford and Johnston, 2007)

	Nitrogen applied in spring (kg/ha)			
Exchangeable K (mg/kg)	120	160	200	240
	Grain Yield (t/ha)			
106	9.66	9.23	9.29	10.33
133	10.80	11.03	10.99	10.94
	N in grain (kg/ha)			
106	146	152	158	178
133	168	184	196	194

There is also a strong interaction between K_{ex}, freshly applied K and N on the yields of grass (Table 7). Permanent grass was grown on two soils with very different amounts of K_{ex}; the largest amount was much larger than necessary but had built up from long-continued application of K fertilizer. On soil with more K_{ex} there was, not surprisingly, no response to fresh K but yield was larger by between 3 and 4 t/ha dry matter with the larger amount of fertilizer N. Yields were much less on soil with the smaller amount of K_{ex} and applying 140 kg K_2O/ha increased yield at both levels of applied N. At both levels of applied N, the smaller yields on soils with little K_{ex} shows that the N was being used inefficiently.

Table 7. Yields of dry grass on permanent grassland on soils with two levels of exchangeable potassium and given two amounts of nitrogen, and four amounts of potash fertilizer. Park Grass, Rothamsted. (Adapted from Johnston et al., 2001b)

Exchangeable K (mg/kg)	N applied per cut (kg/ha)	K_2O applied, kg/ha, each year 0	140	270	540
		Yield, dry matter (t/ha)			
80	40	5.8	6.9	6.8	6.7
	80	6.5	8.3	8.6	8.0
670	40	8.5	9.0	8.6	8.2
	80	11.8	12.0	12.1	12.2

Potassium and grass production

Luxury uptake of K by grass

In the 1950s-1970s much was written about luxury uptake of K by grass and this was often related to the risk of hypomagnesaemia in grazing cattle. However, luxury uptake of K needs to be questioned. The possibility of luxury uptake came about when grass dry matter yield was plotted against percent K in dry matter. Frequently, yield reached a maximum at about 2.2-2.4% K in dry matter, and concentrations above this were considered to indicate luxury uptake of K, i.e. above the needs of the crop. However, if yield is being increased by the application of N, and N application is the driver of cell division and cell enlargement, then there will be more water in the crop. If there is more water in the crop then more K is required to maintain turgor, thus K is being taken up to fulfil a very important requirement in the functioning of the crop and this is not luxury uptake of K. Work with arable crops shows that K in the "plant preferred" cation to maintain turgor. In its role of turgor generation, K can be replaced by other cations, like calcium, sodium and magnesium when there is too little plant-available K in the soil. Thus for grass required for animal feed it would be useful to see to what extent the K required to maintain turgor could be replaced by magnesium (Mg) to achieve the necessary dietary intake, or sodium (Na), which appears to make the grass more palatable for cattle. However, if for grass, K is the "plant preferred" cation then applying a Mg-containing material to a grazed pasture would be one way of ensuring that there was sufficient Mg available to grazing animals.

Potassium in grass and animal health

It has frequently been stated that with intensively managed grassland, applying large amounts of K, especially in spring, can lower the concentration of magnesium

(Mg) in the herbage with the increased risk of hypomagnesaemia (grass tetany) in grazing animals. No farmer wants his animals exposed to the risk of hypomagnesaemia. However, there are some issues about the link between the application of K fertilizer and sufficiently low concentrations of Mg in the herbage to cause hypomagnesaemia in grazing cattle. Hemingway and Parkins (2001) critically reviewed much of the early work on hypomagnesaemia and these and other data were summarised by Hemingway (2005). It was pointed out that in the earliest experiments in England in the 1950s, attempts were made to find fertilizer regimes that might lead to severe hypomagnesaemia so that the condition could be studied and various methods of Mg supplementation devised. All the early investigations involved only three or four cows in mid to late lactation with milk yields generally below 25 litres per day, so that it was impossible to make a statistical evaluation of the data. These authors contend that some of the very large inputs of N and K were unrealistic relative to applications that would be made in normal, on-farm practice. The early experiments also showed that application of Mg-containing materials to the pastures was effective in preventing hypomagnesaemia. However, Hemingway (2005) also noted the well-publicised recommendations included minimising the spreading of slurry or potassium fertilizers prior to spring grazing and the use of repeated light dressings of K fertilizer. He did also draw attention to the need to replace the large amounts of K removed in large yields of silage and hay but commented that these should be applied to silage and hay crops and not to grazing land.

Jarvis and Fisher (2007) note that there has been much discussion with regard to K and its role in influencing the metabolic well being of ruminants. It is clear that ingested K is often in excess of the amount required by the animal, but the impact of that excess is not always clear cut. However, it is generally accepted that high levels of dietary K can have a strong negative impact especially on dairy cow health (Cherney et al., 1998). High dietary K can have two major effects. First, it can increase the risk of hypomagnesaemia resulting from a deficiency in the availability of dietary Mg, particularly when animals graze early sward growth in spring. Second, it may result in hypocalcaemia (milk fever) resulting from a deficiency of dietary calcium (Ca) at the onset of lactation with probably not too much effect during lactation itself. Jarvis and Fisher (2007) note also that the interactions that occur between K and other important dietary nutrients are complex and occur at all stages of the transfer of nutrients along the pathway:

SOIL availability --- PLANT uptake --- ANIMAL ingestion --- RUMEN utilisation

Interactions between these different stages in providing nutrients for animals are frequently influenced by other external factors. Jarvis and Fisher (2007) conclude “that the main cause of dietary Mg problems is too little Mg in the total diet which can be influenced by a high level of K uptake by grasses which, in turn, can be better regulated by a greater degree of control of K supplies from soil, manures and fertilizers."

Clearly there is much evidence that Mg supplements work and perhaps the best is application of Mg-containing material to the grazed sward so that Mg is equally available to all grazing animals. The statement about controlling K supplies is perhaps a little too simplistic. Based on the N x K interactions discussed earlier, the application of N encourages cell division and expansion, an increase in the water content of the crop and consequently increased uptake of K as an osmoticum to maintain turgor. On this basis it would seem to be more important to control N supply to regulate growth and the consequent uptake of K, rather than the availability of K *per se.*

Table 8. Soil potassium balance for grazed grass on a drained silty clay soil in SW England with low input and high input systems. (Mean of two years, adapted from Jarvis and Fisher, 2007)

		"Low input"	"High input"
		no N, 50 kg/ha K	FYM+280 kg/ha N + 149 kg/ha K
Inputs (kg/ha)		kg/ha K	
	Rainfall	8	8
	Fertilizer	50	149
	FYM	none	213
	Recycled urine	35	59
	Recycled dung	8	15
	Total	101	444
Outputs (kg/ha)			
	Leaching	7	15
	Grazing	87	106
	Silage	none	371
	Animals	1	2
	Total	95	494
Average balance		6	-50
Range		(+22, -9)	(-77, -24)

Jarvis and Fisher (2007) pointed out that most grassland soils will require inputs of K as fertilizer or slurry to maintain plant-available soil K at the level required to produce the large yields of grass needed to sustain a high level of animal production. As noted in the paragraph above, however, while N supply controls yields, it also influences the uptake of K. These authors suggest the need to develop at the field level a K balance and budget for grassland. A comparison of the K budget for two grazed fields, one intensively and the other extensively managed, on a drained silty clay soil is shown in Table 8.

The data illustrate a number of points. 1. Although the total K input to the soil in the intensive system was more than four times that in the extensive system the soil balance was still negative, on average 50 kg/ha over two years. 2. There was considerable year to year variation – such budgeting needs to continue for a number of years. 3. Large amounts of K are involved even in the low input system. The differences in K balance have a significant effect on the distribution of K in the different soil pools (Table 9). The negative K balance in the intensively managed system, even with the large input of K, decreased the amount of K in the soil solution and readily available pool, which is the K most accessible to plants, by about 25% and 57%, respectively, compared with the low input, extensively managed system. Such decreases will eventually result in too little soil K being available to support economically viable yields of crops.

Table 9. Effect of potassium balance on soil K pools in "low input" and "high input" grazed grass systems[1] on a drained silty clay soil in SW England. (Adapted from Jarvis and Fisher, 2007)

	K pool[2]		
Factor	Soil solution K (mg/kg)	Exchangeable K (mg/kg)	Non-exchangeable K (mg/kg)
Low input[1]	24[a]	173[a]	2662[a]
High input	18[b]	99[b]	2763[a]
Soil undrained	26[a]	174[a]	2740[a]
Soil drained	16[b]	98[b]	2686[a]

[1]low and high input as in Table 8

[2]pools as described in text and shown in Fig.1

Different letters in columns indicate significant differences ($P<0.05$)

References

Ahmad, N. and Wyn Jones, R.G. (1982). Tissue distribution of glycinebetaine, proline and inorganic ions in barley at different times during the plant growth cycle. *Journal of Plant Nutrition.* 5, 195-205.

Barraclough, P.B. and Leigh, R.A. (1993). Grass yield in relation to potassium supply and the concentration of cations in tissue water. *Journal of Agricultural Science, (Cambridge).* 121, 157-168.

Cherney, J.H., Cherney, D.J.R. and Bruulsema, T.W. (1998). Potassium management. In: J.H. Cherney and D.J.R. Cherney (eds) *Grass for Dairy Cattle*. CABI Publishing, Wallingford, UK. pp 137-160.

Hemingway, R.G. (2005). Fertilisers and Hypomagnesaemia: a Historic Exaggeration? The Potash Development Association, York, UK. 12 pp.

Hemingway, R.G. and Parkins, J.J. (2001). Fertiliser usage and the mineral requirements of grazing livestock. Proceedings 466. International Fertiliser Society, York, UK. 36 pp.

Jarvis, S.C. and Fisher, G.E.J. (2007) Potassium, sodium and magnesium in grass and animal nutrition: a question of balance. Proceedings 618a International Fertiliser Society, York, UK. 18 pp.

Johnston, A.E., Poulton, P.R. and Syers, J.K. (2001a) Phosphorus, potassium and sulphur cycles in agricultural soils. Proceedings 465. International Fertiliser Society, York, UK. pp 2-44.

Johnston, A.E., Poulton, P.R., Dawson, C.J. and Crawley, M.J. (2001b). Inputs of nutrients and lime for the maintenance of fertility of grassland soils. Proceedings 486, International Fertiliser Society, York, UK. 40 pp.

Krauss, A. (1997). Plant nutrition and biotic stress, In: Johnston, A.E. (ed) *Food Security in the WANA Region, the Essential Need for Balanced Fertilization*. International Potash Institute, Horgen, Switzerland. pp.295-304.

Leigh, R.A. and Johnston, A.E. (1983a) Concentrations of potassium in the dry matter and tissue water of field-grown spring barley and their relationships to grain yield. *Journal of Agricultural Science, (Cambridge)*. 101, 675-685.

Leigh, R.A. and Johnston, A.E. (1983b). The effects of fertilisers and drought on the concentrations of potassium in the dry matter and tissue water of field-grown spring. *Journal of Agricultural Science, (Cambridge)*. 101, 741-748.

Marschner, H. (1995). *Mineral Nutrition of Higher Plants*. 2nd Ed. Academic Press, London. 889 pp.

Mengel, K. (1997). Impact of potassium on crop yield and quality with regard to economical and ecological aspects. In Johnston, A.E. (ed) *Food Security in the WANA Region, the Essential Need for Balanced Fertilization*. International Potash Institute, Horgen, Switzerland. pp. 157-174.

Milford, G.F.J. and Johnston, A.E. (2007). Potassium and nitrogen interactions in crop production. Proceedings 615, International Fertiliser Society, York, UK. 24 pp.

Wiebel, J. (1997). Potassium and the nutritional value of fruits and vegetables. In: Johnston, A.E. (ed) *Food Security in the WANA Region, the Essential Need for Balanced Fertilization.* International Potash Institute, Horgen, Switzerland. pp. 224-238.

Wyn Jones, R.G., Brady, C.J. and Spiers, J. (1979). Ionic and osmotic relations in plant cells. In: D.L. Laidman and R.G. Wyn Jones (eds) *Recent Advances in the Biochemistry of Cereals*. Academic Press, London. pp 63-103.

The Agricultural Catchments Programme – Measuring Change on Irish Farms

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Programme Overview

Innovation is central to the design and delivery of the Agricultural Catchments Programme (ACP). It is a national programme working with farmers and other stakeholders to generate new knowledge and quickly disseminate it to farmers and the wider community. What is new and most exciting about the programme is that it is being delivered by an integrated team of researchers and advisers working closely with farmers to ensure the free flow of new knowledge and ideas among all the participants. The team also has close links with other Teagasc researchers and advisers and collaborates with other agencies. By facilitating this sharing of information and experience, the programme can accelerate knowledge transfer and maximise its impact.

The Programme aims to support farmers in handing on a legacy of sustainable farms to the next generation. To be truly sustainable these farms must be able to operate on a sound socio-economic basis while ensuring that their activities are compatible with the maintenance of waters at good quality or its improvement above this level. The drivers of this programme can, therefore, be categorised as both socio-economic and environmental.

Socio-economic Drivers

Agriculture and food is an important indigenous sector of the Irish economy and accounts for 6.3% of GDP, 8.2% of employment and 10.5% of total exports. It is in Ireland's interest that we not only maintain the sector's current contribution to the economy but be in a position to increase output when the opportunity arises; for example by increasing milk production if quotas are abolished. At catchment level this means helping participating farmers to achieve long-term economic sustainability by working in partnership with them to develop a good understanding of their businesses and developing them through consultation and farm-planning.

Environmental Drivers

The Nitrates and Water Framework Directives are the main environmental drivers of the ACP. The Nitrates Directive (ND) aims to minimise nutrient losses to water bodies from agricultural sources and is based on managing the rate, timing and accumulation of nutrients to avoid excessive or untimely transfer to water. Ireland's National Action Programme (NAP) under the ND is put into law by S.I. No. 101 of 2009 - Good Agricultural Practice (GAP) for Protection of Waters Regulations. These regulations cover a broad range of farm practices and include limits on nitrogen and phosphorus applications (both artificial and in manures), closed periods for fertilizer spreading, minimum manure storage requirements and restrictions on winter ploughing. This single set of measures covers the whole country with only minor variations from region to region.

Article 5(6) of the ND states that *"Member States shall draw up and implement suitable monitoring programmes to assess the effectiveness of action programmes"*. The ACP was put in place in response to this requirement and to fulfil Ireland's obligation for a monitoring programme at the catchment scale.

The Water Framework Directive (WFD) brings together a range of EU directives which relate to water management, including the ND, and combines chemical and ecological standards for water quality. A fundamental WFD objective is to maintain the "high status" of waters where they exist, prevent any deterioration in the existing status of waters, and achieve at least "good status" in relation to all waters by 2015. This objective is very broad in its scope and relates to water quality in rivers, lakes, canals, groundwater, transitional (estuarine) waters and coastal waters out a distance of at least one nautical mile. The WFD is therefore a significantly more integrated and ecologically based Directive with more stringent targets, based on ecological rather than chemical criteria, compared with the ND.

Member States have to ensure that a co-ordinated approach is adopted for the implementation of programmes of measures for the achievement of the WFD objectives. It clearly links water quality with land management activities including those associated with agriculture. In this context the ACP will also consider the WFD water quality targets and contribute to evaluating any future impact of additional measures introduced under the WFD.

Programme Objectives

The ACP is funded by the Department of Agriculture, Fisheries and Food (DAFF). When DAFF made the original request to Teagasc to prepare an Operations Action Plan for the establishment of an agricultural catchment monitoring programme it outlined the broad objectives for the programme:

- To establish baseline information on agriculture in relation to both the ND and the WFD.

- To provide an evaluation of the GAP measures and the derogation in terms of water quality and farm practices.
- To provide a basis for a scientific review of GAP measures with a view to adopting modifications where necessary.
- To provide better knowledge of the factors which determine farmers' understanding and implementation of the GAP.
- To provide national focal points for technology transfer and education for all stakeholders in relation to diffuse nutrient loss from agriculture to water.
- To include monitoring which may be necessary for the purposes of the WFD.

Challenges

The major scientific challenge for the programme is to provide the evidence on which the NAP (and subsequently WFD) measures can be evaluated. The basis of this will be the reduction in the nitrogen and phosphorus leaving agricultural land, transported to water and possible impacts on its quality. If there are indications that water quality targets are not being achieved due to GAP then modifications to measures will need to be considered.

A second major challenge is the integration of environmental monitoring with profitable farming to provide models for more sustainable farming systems. The ACP approach to this challenge involves the farming community, advisers, researchers and the other stakeholders in the selected catchments. The participation of the farming community is the cornerstone of the programme. The context of their contribution is against the background of significant changes occurring in agriculture and the legitimate concerns regarding the implications of the NAP measures on their activities and profitability. From the outset the farm organisations have been involved in the programme at national level through the Consultation and Implementation Group, and at local level farmers are involved in the programme through one-to-one consultation and in groups. Through these mechanisms, the Programme will generate evidence of the farming community's compliance with the measures, the linkages between nutrient flows on the farm and water quality, the impact of the measures on farm profitability, and a greater insight into socio-economic barriers to their adoption. Heretofore, these linkages often have been implied or misunderstood causing legitimate concern in the agricultural community.

The Programme Approach

A unique strength of this programme is that production, financial, nutrient and socio-economic data from individual farms will be collected. Evidence from the programme will demonstrate farmers' compliance with the measures. Advisers, researchers and the other stakeholders will work with farmers to achieve the best outcome from the programme and it will provide the basis for the broad range of research and advisory work required to develop sustainable farming in Ireland over the next two decades.

Catchment Selection

As advised under EU guidelines, to represent the range of farming types and intensities, and physiographic differences in the country, a GIS multi-criteria decision analysis (MCDA) was used to select a series of catchments that maximised agricultural intensity and minimised non-agricultural land uses. The scale most suited to the ACP was considered to be $4km^2$ to $12km^2$ to ensure a range of hydrology from headwaters to main channel but also small enough to engage with all of the farming stakeholders in each catchment. Four catchments have been selected with this process and a further karst catchment was selected by analysing selected GIS data layers. A number of derogation holdings are also included within the catchments. The catchments range in terms of agricultural intensity, land use (grassland or arable) and of N and P loss to streams (vertical drainage or lateral runoff risk, respectively).

Advisory Service

Advisers and technicians work in teams of two to support the farmers in the catchments. Each adviser works closely with approximately 40 farmers who farm the bulk of the land in the catchments. This adviser/technician team collect production, environmental and financial data and, with the farmer's agreement, the advisers prepare farm management plans which will include both production and financial targets. Thus participating farmers have the support of a comprehensive advisory package covering financial, production and environmental issues. Other farmers who have small amounts of land in the catchments, or land on the periphery, also receive advisory support primarily focused on nutrient management, and basic nutrient input and output data is gathered on these farms.

Experimental Design

The scope of the ACP experimental design is related to providing a baseline during the early years of the GAP regulations and to monitor nutrient transfer trajectories based on the nutrient transfer continuum concept (Haygarth et al., 2005). In this model, nutrient *sources* as inputs at the farm scale and field soils are exposed to a *mobilisation* mechanism via hydrological *pathways* and are *delivered* to streams or other water bodies where a eutrophic *impact* may manifest. National standards for monitoring nutrient transfer and potential trophic impact via both Nitrates and Water Framework Directive regulations are in source (nutrient N and P use; soil P status), delivery (chemistry standards for N and P in rivers and groundwater) and impact elements (drinking water N standards, especially in groundwater; Ecological Quality Ratios in rivers). That is to say, there are no standards for mobilisation or pathway elements (other than when groundwater layers are considered as pathways).

Using the continuum concept, use and source elements are being audited, as far as possible, on a 100% coverage basis - on a field by field basis. Soil nutrient status (P (Morgan available P), K, pH (for lime requirement)) is also being audited on a field-by-field basis (or ≤2ha basis, whichever is the smallest) and nutrient management plans prepared for core catchment farmers. These data will provide information on both nutrient use compliance and also the status of soils (for P). One of the major policy drivers here is to return high P soils to at least an index upper boundary of 3 (8mg L^{-1} Morgan P for grassland and 10mg L-1 for arable). Also in each catchment, a characterisation of soil P chemistry and dynamics in the 0-10cm zone, in addition to total P, will include information on P desorbability (soil P saturation) and soil dispersion (for erodibility) for comparisons within and between catchments.

At the delivery end of the continuum, as nutrients cascade through the system from current and prior management, the ACP is ambitiously monitoring water quantity and water quality on a continuous basis at the outlets of the catchments. In other studies, continuous water quantity has been allied with coarser resolution water quality sampling and examples (Johnes, 2007; Bechmann and Stålnacke, 2005) show how uncertainty from coarse resolution samples can lead to flawed inter-annual nutrient load comparisons and that simple load comparisons can also be influenced by variations in inter-annual rainfall and runoff relationships. As Irish catchments tend not to be 100% agricultural, it is also important to be able to highlight the role of rural point sources, such as small waste water treatment plants and/or septic systems. Total P, total reactive P, total oxidised nitrogen (TON = NO_3^- + NO_2^-) are monitored on a sub-hourly basis with water discharge. Measurements of turbidity (suspended sediment) and conductivity are also monitored to provide information relating to sediment associated nutrient flux and pollution spikes. Additional ‘snap-shot’ N and P monitoring sites at other tributary and ditch locations in each catchment upstream of the outlets is also providing data on water from different parts of the hydrological cycle where the influence of shallow and deep groundwater, storm water etc. can be important at different times.

These source start and delivery end point audits are important parts of the ACP experimental design; it will be important to demonstrate compliance in terms of source use but also to be able to concentrate management advice in those field parcels that have a legacy of intensive fertilisation and where that legacy might, for example, have resulted in very high soil P concentrations and/or accumulation of N in slow, subsurface flowpaths. Monitoring both start and end points will, for example, be important if compliance (soil P, P and N use) and/or trajectory towards compliance (e.g. soil P status) is evident, but that other factors such as annual climatic variations, rural point sources or historical management, overly influences the trajectory of water quality response (Kleinman et al., 2007).

Due to the variable and dispersed nature of hydrological flow pathways, it will not be possible to monitor these on a 100% basis. Therefore, following conceptualisation of the major flow pathways, demonstration pathway studies are supplementing the source and delivery audits. Where N sub-surface flow is regarded as the most significant flow pathway, nested groundwater bores have been installed (two transects of three bores – three levels in each bore – i.e. 18 levels in each catchment). The experimental design will therefore demonstrate N transfers in the saturated zone and the bores will act as infrastructure to monitor N (and P) in this pathway as well as points for comparing against groundwater nitrate standards.

For demonstrating surface and near surface pathways, small subcatchments with different connectivity characteristics will be monitored. This measurement of connectivity is defined by high resolution digital elevation models (DEMS) based on topography. This approach avoids the establishment of field plots but accommodates catchment hydrology and land use processes. In conjunction with source data, these sub-catchments can be delineated in terms of, for example, high connectivity and high nutrient status; low connectivity and low nutrient status; and levels in between. The experimental design will test these assumptions using targeted storm sampling that can be compared with the nutrient results delivered to the catchment outlets.

The links between diffuse nutrient transfer in rivers and their ecological status is still not fully understood; however, it will be important to characterise ACP rivers according to ecological standards currently used in Ireland. Here, a number of direct and indirect ecological metrics are being used to assess and characterise the rivers and provide an extended baseline. These include Q-values, BMWP/ASPT metrics, Trophic Diatom Index, River Habitat Assessment Tool (for hydromorphology) and an assessment of fish populations.

Socio-Economic

Economic – Micro level farm data will be used in conjunction with parametric (stochastic frontier analysis) and non-parametric (data envelopment analysis) estimation techniques to assess the nutrient management efficiency and economic performance of all farmers in the selected river catchments areas (Coelli et al., 1998). Application of these techniques will allow economic benchmarking of farms nationally in each catchment with the most efficient farms of a similar size and system to establish if nutrients are being managed and applied efficiently.

Attitudinal – A mix of quantitative and qualitative techniques will be employed to assess farmer attitude to adoption of farm practices that could promote improved water quality status. The decision of a farmer to adopt and undertake environmentally desirable farm practices will be contextualised as a human behavioural issue. A model from the social psychology literature named the theory of planned behaviour (TPB) will be applied (Ajzen, 2005). According to the TPB, intention is based on three main constructs, namely: attitudes, subjective norm (SN) and perceived behavioural control (PBC). Intention to perform a behaviour is considered the most important immediate determinant of that action (Ajzen, 2005). The principle objective is to identify the factors that drive the intention to perform a behaviour, in this instance nutrient management best practice.

In conjunction with the TPB, Q methodology will be employed to evaluate farmers' subjective opinions of water quality policy measures and for the environment generally. As set out by Hall (2008), Q methodology involves a number of stages including identifying the area of discourse and the relevant population, collecting a range of statements relating to the discourse, and selection of a limited number of representative statements from all of those collected. Participants are then required to rank or 'sort' the statements against a scale (usually agree to disagree). Statistical analysis of the 'sorts' is then carried out to enable the extraction of a few 'typical' sorts. Finally, these typical sorts are described and interpreted (Barry and Proops, 1999).

Possible Impacts of ACP Findings

Under Article 27(1) of SI 101 of 2009 the Minister for Agriculture, Fisheries and Food is given the responsibility for monitoring the effectiveness of the NAP measures, including the nitrates derogation. The ACP is the principle means of doing this and the outputs of the Programme will be a main source of scientific evidence of the effectiveness of the measures. This evidence will be important in the formulation of Irish agricultural policy with regard to EU requirements under the Water Framework Directive, of which the Nitrates Directive is a Basic Measure. In the absence of evidence from the ACP, policy-makers would have to rely on data from other studies outside Ireland and from studies conducted at plot or lysimeter scale.

Catchment studies, in general, will deliver their most valuable output over the medium to long term. The implementation of the NAP measures has brought about changes in farm practices on Irish farms which are designed to reduce the risk of nutrient loss to waters. Given the nature of most Irish soils, it is anticipated that there will be a considerable lag-time between the introduction of new practices and reductions in risks to waters. It is likely that this lag-time concept will feature prominently in making the case for further extensions of Ireland's current NAP and Derogation. The ACP's physical infrastructure and experimental design has been conceived to enable it to effectively monitor such subtle medium to long-term effects and demonstrate the impact of this lag-time at catchment scale using a group of real farms being managed in compliance with the NAP measures. The same kind of lag-time effect is likely to occur with regard to socio-economic changes.

References

Agri-vision 2015 committee Report (www.agri-vision2015.ie): Rural Ireland 2025 – Foresight Perspectives published by NUI Maynooth, Teagasc and UCD.

Bechmann, M. and Stålnacke, P. (2005). Effects of policy induced measures on suspended sediments and TP concentrations from three Norwegian agricultural catchments. *Science of the Total Environment*, 304, 238-250.

Central Statistics Office, (2008). Output, Input and Income in Agriculture 2008 – Final Estimate. www.cso.ie/releasespublications/documents/agriculture/current/

Draft guidelines for the monitoring required under the Nitrates Directive (91/676/EEC). Version 3, with annexes 1 through 3. European Commission, Directorate-General XI (Environment, Nuclear Safety and Civil Protection) Directorate D (Environment, Quality and Natural Resources).

EC, 2007. Commission Decision 2007/697/EC of 22 October 2007 granting a derogation requested by Ireland pursuant to Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources.

European Water Framework Directive. www.wfdireland.ie/body wfd.htm

Fealy, R.M., Buckley, C., Mechan, S., Melland, A., Mellander, P.-E., Shortle, G., Wall, D. and Jordan, P. The Irish Agricultural Catchments Programme: catchment selection using spatial multi-criteria decision analysis. Submitted to *Soil Use and Management.*

Haygarth, P.M., Condron, L.M., Heathwaite, A.L., Turner, B.L. and Harris, G.P. (2005). The phosphorus transfer continuum: linking source to impact with an interdisciplinary and multi-scaled approach. *Science of the Total Environment*, 344, 5-14.

Johnes, P. J. (2007). Uncertainties in annual riverine phosphorus load estimation: impact of load estimation methodology, sampling frequency, baseflow index and catchment population density *Journal of Hydrology*, 332, 241-258.

Jordan, P., Arnscheidt, J., McGrogan, H. and McCormick, S. (2007). Characterising phosphorus transfers in rural catchments using a continuous bank-side analyser. *Hydrology and Earth Systems Science*, 11, 372-381.

Kleinman, P. et al. (plus 7 co-authors) (2007). Dynamics of P transfers from heavily manured coastal plain soils to drainage ditches. *Journal of Soil and Water Conservation*, 62, 225-235.

Lane, S.N., Brookes, C.J., Heathwaite, A.L. and Reaney, S.M. (2006). Surveillant science: challenges for the management of rural environments emerging from the new generation diffuse pollution models; *Journal of Agricultural Economics*, 57, 239-257

Schulte, R., Melland, A., Fenton, O., Herlihy, M., Richards, K., and Jordan, P. Modelling soil phosphorus decline; expectations of Water Framework Directive policies in Ireland. Submitted to *Environmental Science and Policy*.

Nitrogen Use Efficiency – Best Management Practices

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Introduction

Agricultural intensification of production from grassland has led to serious imbalances between inputs of N (in purchased fertiliser, feeds and atmospheric deposition) and outputs (mainly milk and meat). Large quantities of N are recycled to land during grazing or application of organic manures. The uptake of N from synthetic fertilisers (e.g. urea, calcium ammonium nitrate) is seldom greater than 50% of the N applied, while N efficiency for global animal production is only 10% (van der Hoek, 1998). The efficiency for ruminants is much lower than for non-ruminants. Average global N efficiencies are 7.7% for the production of cattle, 6.2% for sheep, 20.5% for pigs and 33.8% for poultry.

One of the reasons for poor N use efficiency is high losses from the soil-plant-animal system. N can be lost by ammonia volatilisation, during nitrification and denitrification, by leaching, erosion and surface runoff. The relative importance of each loss process depends on the agricultural system, climatic and edaphic factors, fertiliser form and the amount, method and timing of fertiliser and manure applications. The loss of N represents an economic loss to farmers and has environmental implications for water quality and greenhouse gas emissions. Environmental economists have assessed the total external environmental and health costs of UK agriculture to be £2343m/yr (Pretty *et al.*, 2000).

Ammonia loss by volatilisation has a marked influence on atmospheric chemistry and the acidity of precipitation. The deposition of NH_3 can cause acidification in poorly buffered soils and can upset the ecological balances and diversities in sensitive ecosystems. The processes of nitrification and denitrification result in the release of oxides of nitrogen, particularly nitric oxide (NO) and nitrous oxide (N_2O) into the atmosphere. Nitrous oxide is a potent greenhouse gas (GHG) and also contributes to the destruction of the ozone in the stratosphere (Crutzen, 1981). Nitrous oxide has a 100-year global warming potential (GWP) which is expressed in the form of CO_2 equivalents (CO_2e). Nitrous oxide is 298 times more potent as a GHG, relative to carbon dioxide (IPCC, 2007). Of the 3.2Mt N_2O-N/yr emitted globally from anthropogenic sources approximately 60% originates from agricultural soils, as a result of nitrification and denitrification. In addition to the direct sources of N_2O, there are also indirect ones that include N re-deposited onto land following NH_3 and NO_X volatilisation (EF 0.010 kg N_2O/kg NH_3-N) and N losses to land drainage water (leaching and /or runoff), where it can be partially transformed to N_2O (EF 0.0075 kg N_2O /kg N leached). Figure 1 shows the range in GHG emissions from agriculture as a % of total national emissions in various countries worldwide. Agriculture accounted for 27% of total Irish GHG emissions in 2007 and nearly 50% in New Zealand, indicating the importance of agriculture in the economy of these two countries.

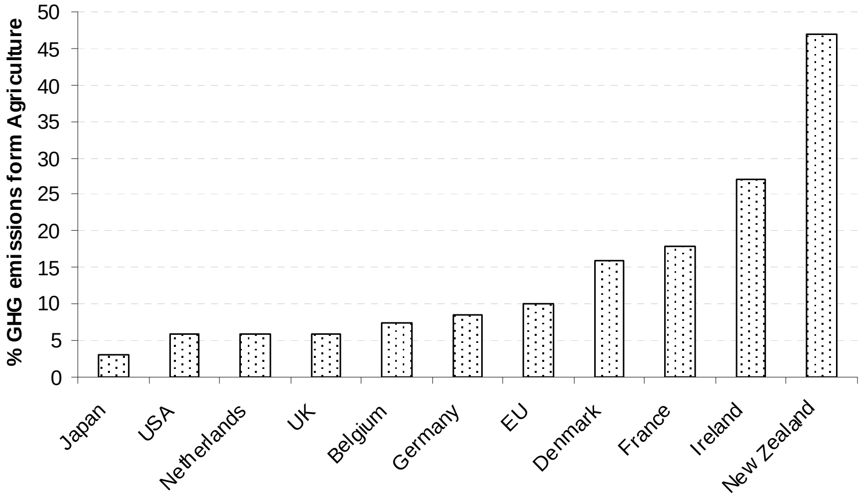

Figure 1. Emissions from agriculture as a percentage of total national emissions of greenhouse gases (source: UNFCCC).

The UN Framework Convention on Climate Change (UNFCCC) and its Kyoto Protocol provide the basis for international action to address climate change. The UK Climate Change Act (2008) requires a reduction in GHG emissions of 80% by 2050 (relative to 1990 levels), whereas the EU Emissions Trading Directive requires member states to reduce emissions by 20% by 2020 (EU, 2009).

The main concerns about N loss to water is compliance with EC directives which define a maximum admissible concentration of 11.3 mg NO_3^--N/l, 0.38 mg NH_4^+-N/l and 30 µg NO_2^--N/l for potable use and the potential role of N in eutrophication of surface waters. The EU Water Framework Directive (EU, 2000) aims to bring together many existing directives related to protecting water quality such as the Groundwater Directive 1980, Nitrates Directive 1991, Drinking Water Directive 1991 and the Bathing Water Directive 2006. It commits member states to restore all water bodies to good ecologic and chemical status by 2015.

Good management strategies to improve N use efficiency are based on matching N supply to crop demand taking account of N supplied by the soil, previous crop residues and organic manures. These management strategies include N fertiliser management, crop and soil management, livestock management, manure management and modification of N fertilisers.

N Fertiliser Management

The Nitrates Directive Action Plan in Northern Ireland prohibits the application of chemical N after 15 September and before 1 February. Within the permitted spreading period, application of fertiliser at the time of maximum crop demand i.e. March to June, results in increased N efficiency and therefore lower N losses.

Calcium ammonium nitrate (CAN) is the main source of manufactured N fertiliser used in Ireland. In Northern Ireland, of the 212 thousand tonnes of fertilisers used in 2009, urea-based N accounts for typically 2.8 % of N use, CAN 50%, and compounds and blends 46% (DARD, 2009). The yield response to urea fertiliser is often lower than that to CAN for many crops due largely to ammonia volatilisation, following rapid hydrolysis of urea by soil urease activity. Factors that affect NH_3 volatilisation include rate of application, soil properties (H^+ buffering capacity, cation exchange capacity, pH, texture and level of Ca^{2+} salts), soil temperature, soil water content (evaporation, rainfall and irrigation) and air exchange rates. Under field conditions the environmental factors of temperature, soil water content and air exchange at the soil surface are overriding factors in determining the magnitude of NH_3 loss. The amount and timing of rainfall after fertiliser application are important factors affecting urea efficiency. Short-term rainfall immediately after urea application can lower NH_3 volatilisation. In field studies ammonia loss was greatly reduced if more than approx 10mm of rain fell within 2 days of fertiliser application or at the start of an intensive grazing period. High NH_3 loss rates are generally associated with periods of soil drying when wet (near field capacity) surface soil is followed by several days with little or no rainfall. However, on soils at or near field capacity, excessive long-term rainfall or an intense rainfall event, immediately after fertiliser application can lead to loss of N (from all N fertilisers) by surface runoff.

Numerous field trials have compared the yield responses of temperate grassland to urea or CAN. In a review of agronomic trials in the British Isles, Watson *et al*. (1990) concluded that, on average, urea was as effective as CAN when applied to grassland in spring, but was less effective in summer. However, more recent studies (2002-2005) in the UK (DEFRA, 2006) with grass and winter cereals have shown high NH_3 volatilisation losses from urea in March-April using wind tunnels. The average NH_3 emission factor (EF) from granular urea was 27% at 15 grassland experiments and 22% at 13 winter cereal experiments. The average NH_3 EF from ammonium nitrate was less than 3%. Research has shown that NH_3 losses from urea-based fertilisers are variable, unpredictable and can be large, depending on climatic and soil factors. On average, the use of solid urea in practice in the UK would need approximately 20% more N to achieve the same cereal crop yield and quality as ammonium nitrate. However, the efficiency of urea can be improved by the addition of a urease inhibitor (discussed later).

The form of N applied has a major effect on N_2O emissions, particularly under wet conditions. AFBI research has compared CAN, urea, urea amended with a urease inhibitor Agrotain (urea + Ag) and urea ammonium sulphate (UAS) applied to grassland on N_2O emissions following applications in spring (April), early summer (May) and mid summer (June) in 2003. When conditions were wet in May, N_2O emissions were up to three times greater from a nitrate containing source (i.e. CAN) compared to NH_4^+-N based fertilisers (Table 1).

Table 1. N_2O emission factors for each fertiliser-N, and seasonal weighted mean EFs. IPCC default emission factor = 1.00% (uncertainty range 0.3 – 3.0%).

		Net N_2O emission factor (%)			
Site/Crop	N material	Spring	Early Summer	Mid Summer	Seasonal weighted mean
Hillsborough (grass)	CAN	0.13	10.99	0.81	3.93 ± 1.17[b]
	Urea	0.06	4.47	0.92	1.74 ± 0.47[a]
	Urea+Ag	0.25	4.63	0.61	1.80 ± 0.48[a]
	UAS	0.07	3.05	0.96	1.29 ± 0.42[a]

[a]Net emission, after subtraction of control value, as % of N applied.
Values with different letters are statistically different ($p<0.05$).

The dominant N fraction lost to drainage water or ground water is nitrate. Nitrate in soil arises from a variety of soil processes of which the turnover of soil organic matter is especially important. Fertiliser N and the grazing animal influence these process rates markedly. Nitrate leaching can be five times greater on grazed swards compared to grassland cut for silage, due primarily to the development of N ‘hotspots’ in the soil as a result of urine patches. The N application rate in a urine patch can be equivalent to 400-2,000 kg N/ha and as this can greatly exceed crop requirement, the excess is an important source of N loss. A major influence on nitrate leaching is the rate of fertiliser N applied. A ten year study in Northern Ireland showed that there was a linear relationship between NO_3^--N leached and fertiliser N applied, with the equivalent of 13% of N applied being lost to drainage water (Figure 2) (Watson *et al.*, 2000). However, for individual years this varied from 5 to 23%, the highest losses occurring after a dry summer.

The rate of fertiliser N applied also has a major influence on gaseous losses by denitrification. AFBI research has shown a linear relationship between total denitrification (N_2O + N_2) and fertiliser N input from grazed grassland swards in Northern Ireland (Watson *et al.*, 1998a). Maximum rates of denitrification generally coincided with applications of fertiliser N, except when the soil moisture content was low. Loss rates as high as 3 kg N/ha/d were measured. Other studies, on lighter soils have shown more of a curvilinear increase in N_2O with N fertiliser rate.

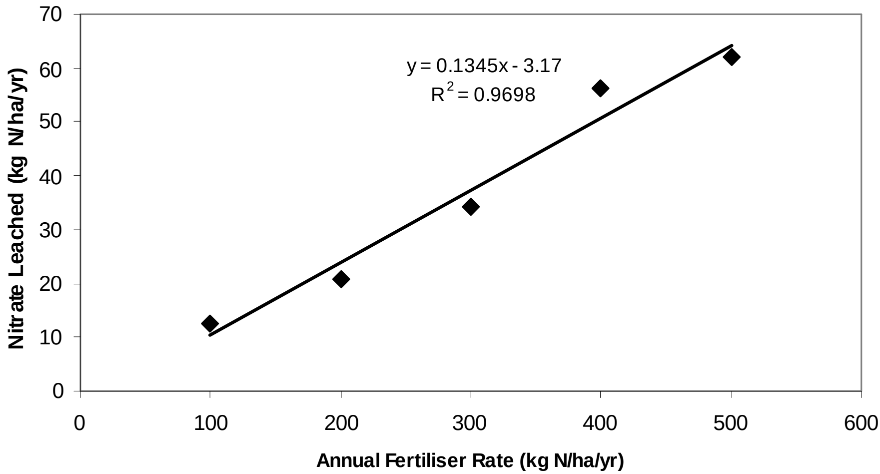

Figure 2. Average nitrate-N load (kg N/ha/yr) lost to drainage water from grazed grassland

Crop and Soil Management

Denitrification is generally high under wet soil conditions, resulting in large N_2O emissions. Drainage of productive lowland areas offers scope to reduce greenhouse gas emissions from agriculture. However, there may be negative effects on soil carbon sequestration and other implications for the rural economy as a whole if this strategy is adopted.

Factors that affect the productivity and N uptake of a grass sward (e.g. age, species, cutting frequency, nutrient balance, soil structure, pH, etc.) may affect N losses through their effect on the quantity of mineral N in the soil. It is important to recognise that the lack of nutrient balance with other essential nutrients (e.g. P, K, S) will reduce N use efficiency and increase overall N losses. Improving soil structure by minimising soil compaction by machinery and/or grazing livestock will improve N use efficiency. Ploughing out of grassland in order to reseed grassland or to sow an arable crop results in enhanced mineralisation of soil organic C and N and may increase N losses. The total amount of N mineralised during the first year after ploughing may exceed 360 kg N/ha, depending on the age of the sward and its recent management (Whitehead 1995). However, ploughing may improve sward quality and N uptake capacity and redistribute nutrients within the plough layer, particularly P, which tends to accumulate in the top few centimetres of soil.

AFBI research has shown that increasing soil pH under laboratory conditions tends to reduce the N_2O mole fraction in the (N_2O + N_2)-N gas products (Stevens *et al.*, 1998). However, more recent studies suggest that this may be due to an increase in

N_2 production rather than a decrease in N_2O production *per se*. Further research is required, particularly under field conditions, to establish if lime has any value as a mitigation strategy to lower N_2O emissions and to improve soil structure. The optimum soil pH for grassland is 6.0 – 6.5, which can be maintained with lime applications. However, excessive use of lime can aggravate trace element deficiencies.

The potential of clover to biologically fix atmospheric N is well recognised, with many studies showing that well managed grass-clover swards can produce similar dry-matter yields to pure grass swards receiving up to 150 kg chemical N/ha. Evidence suggests that the use of clover in grass swards, to partially replace fertiliser N, may lower GHG emissions. However, nitrate leaching from grazed grass-clover swards is similar to grass only swards receiving the same input of fertiliser N as that biologically fixed by clover (Jarvis *et al*., 1996).

Livestock production

When the production per animal is increased, fewer animals are required to maintain the production of the farming system. Although the GHG emission per animal may be higher for a high-producing animal than for a low-producing animal, the effect on GHG emissions will be outweighed by the decrease in the number of animals required to obtain the desired total production. Restricted grazing may be an option to mitigate N_2O emission and NO_3^--N leaching from intensively managed grassland. Grazing animals produce 'hotspots' of N in urine patches (up to 1,000 kgN/ha for cattle) which is well in excess of crop requirements so the excess is lost to the environment. When grazing is restricted, the animals will be housed for a longer time and their excreta will be collected and stored as slurry. This slurry can then be applied uniformly to grassland using low trajectory spreading techniques and consequently less N fertiliser will be required. Nitrous oxide emissions from grassland following grazing are significantly higher than that from stored slurry applied to grass using appropriate spreading techniques. However, there are significant practical issues associated with restricted grazing, including cost and health and welfare implications for the animals.

The distribution of N between dung and urine is dependent on the N content of the herbage. For example, the proportion of excreted N in cattle urine increases from approximately 45% with 1.5% N in the diet, to about 80% when the diet contains 4% N (Whitehead, 1995). This distribution has environmental significance, as most N loss is associated with urine patches. Strategies aimed at reducing the total N intake of the animal, without affecting the nutritional value of the feed will reduce N excretion and therefore lower N losses. These strategies include reducing the diet crude protein concentration, feeding higher amounts of readily fermentable carbohydrates e.g. maize silage and using ‘high sugar’ ryegrass varieties in grazing systems.

Manure management

Approximately 80 % of the manure produced by cattle in Ireland is managed as slurry (Hyde & Carton, 2005). Cattle slurry has been seen as a waste and farmers have not taken into account its valuable nutrient content, until recently. Legislation introduced in Ireland to comply with the EU Nitrates Directive 91/676/EEC requires a closed period for slurry spreading and limits placed on the amount of chemical N that can be applied. There is therefore increased interest in using manure N more efficiently. Technical information ranging from ‘hard copy’ reference books and leaflets through to computer-based decision support systems is now available to farmers, to ensure that fertiliser recommendations are adjusted to take account of the nutrients in organic manures.

Table 2. The magnitude and pattern of ammonia losses after the surface application of cattle slurry to grassland soils (Stevens & Laughlin, 1997).

Study	Time of experiment	Duration (days)	Soil type	NH_3 loss (% NH_4^+-N applied)	Pattern of loss (% of total NH_3-N)
Bussink *et al*., 1994	May-July	10	clay	93	89% in 1day
	May-Sept	4	Peat	52	85% in 1day
	June	4	Sand	61	92% in 1day
Klarenbeck *et al*., 1993	Sept	4	Peat	55	80% in 1day
Lockyer *et al*., 1989	May-June	3-5	Sand	39	>80% in 2day
Moal *et al*., 1995	March	4	Sand	75	75% in 5 hours
			Sand	54	75% in 12h
Pain & Thompson, 1989	April-June	5	Sand	35	40% in 6 hours
					70% in 1day
Pain *et al*., 1990	May-June	3	Sand	39	24-39% in 1hour
					85% in 12 hours

When slurry is surface spread on grassland the fluxes of NH_3 and N_2O are highly variable and are dependent on slurry properties, soil properties and environmental conditions. Most of the NH_3 losses occur in the first few days after application of slurry (Table 2).

In Ireland most of the manure is spread by splash plate and much of the available N is lost to the atmosphere (Table 2). Ammonia losses can be lowered by reducing slurry to air contact time by applying cattle slurry in bands on the soil surface, or by using a trailing shoe. The trailing hose may hang loosely or can be held rigidly with a small metal shoe which rides along the soil surface, parting the herbage (10-

15 cm high) and making sure that the manure is applied directly on to the soil surface in a band. The standing foliage helps to absorb odour and ammonia. At AFBI-Hillsborough a three year study has been conducted to determine the yield response and N efficiency of cattle slurry applied to grassland by splash plate, band spread and trailing shoe methods of application (Figure 3). Overall the band spread and trailing shoe methods resulted in increased dry matter yields of 18% and 26% respectively, relative to splash-plate. Frost *et al.* (2007) showed a potential chemical N 'sparing' effect of up to 44 kgN/ha for cattle slurry applied (mean 55t/ha) using a trailing shoe application method compared to a conventional splash plate system. Low trajectory spreading techniques reduce sward contamination and allow grazing to occur within a few days of application, rather than the normal 3-4 weeks grazing exclusion following surface spreading. Further benefits are reduced odour nuisance and lower risk of airborne transmission of diseases. However, band spreading or trailing shoe technology costs more than a conventional splash plate tanker. There is also a small increase in spreading time and higher maintenance costs associated with the low trajectory spreading equipment.

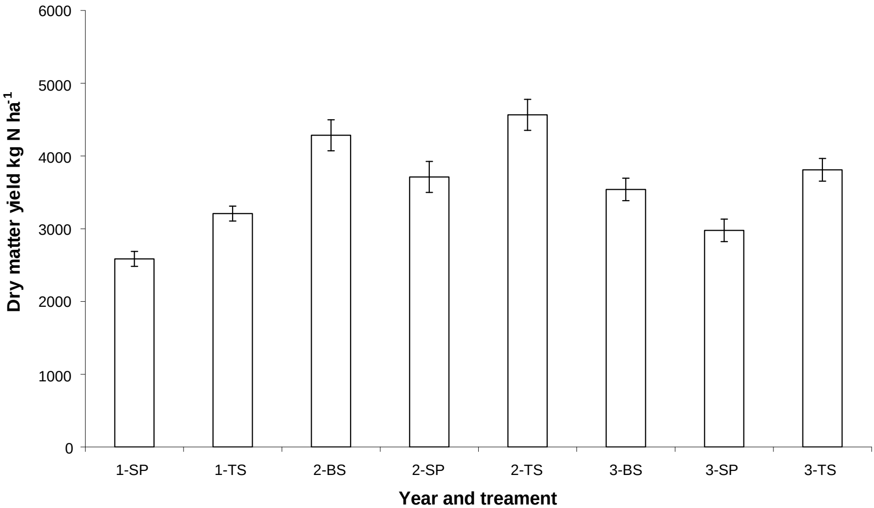

Figure 3. Effect of slurry application method on dry matter yield for each year of experiment. Error bars are ± standard error of the mean. SP=splash plate, TS =trailing shoe and BS=band spreading.

The timing of cattle slurry application is critical for N use efficiency. In Ireland the N fertiliser replacement value of slurry applied with splash plate was 0.25 if applied in spring and 0.05 if applied in summer (Coulter, 2004). In the UK, values for surface applied slurry at 6% dry-matter, are 0.35 for spring and 0.20 for summer (MAFF, 2000). In Ireland it is estimated that 34% of cattle slurry is applied in spring and the remainder in summer months. Application in spring is the most efficient time because there is rapid uptake of N by the sward and N losses by NH_3 volatilisation and leaching are low (Carton & Magette, 1999).

On a dairy farm it is estimated that 25 to 30% of NH_3 emission occurs from stalls and during slurry storage. To reduce NH_3 emissions from stalls, an option is to remove dung and faeces as quickly as possible to enclosed storage. Washing stalls with water and dilute acid can further increase N efficiency (Korevaar and Boer, 1990). In Ireland to comply with EU legislation slurry has to be stored from October to January. Covering of the slurry store can reduce NH_3 emission by 50 to 90%. Formation of a crust on stored slurry by adding chopped straw will also reduce volatilisation by 50 to 70% (Korevaar and Boer, 1990).

The composition of animal slurries in farm storage tanks is variable. Therefore it is difficult to estimate its fertiliser value accurately before spreading. Chemical analyses for NH_4^+-N using steam distillation are accurate but they take time, are costly and require a laboratory environment. However there are fast simple tests for determining the NH_4^+-N concentration in slurries at the farm level. Quick tests include a hydrometer (Tunney, 1984), electrical conductivity meter (Stevens *et al.*, 1995), an Agros N meter and Quantofix N-volumeter (van Kessel and Reeves III, 2000). The NH_4^+-N content of 107 dairy slurries determined by the quick tests was regressed against NH_4^+-N determined by laboratory methods (van Kessel and Reeves III, 2000). Parameters for the regression equations, describing the relationship between the quick tests and laboratory analysis of NH_4^+-N, are shown in Table 3. The strongest relationship was with the Quantofix-N-Volumeter, followed by the Electrical Conductivity meter and the Agros-N-Meter. The data indicate that a number of quick tests are suitable for determining the NH_4^+-N content in dairy cattle slurries. Recent research has demonstrated both the potential of near-infra-red spectroscopy (NIRS) for nutrient analysis of solid manures and the development of a sample homogenisation procedure for much improved consistency of analytical results (Smith *et al.*, 2008a).

Table 3. Parameters that describe the relationship between quick test analysis of manures and standard laboratory analysis (van Kessel and Reeves III, 2000).

Quick test	No. samples	Intercept	$SE_{intercept}$	Slope	SE_{slope}	r^2
Hydrometer	105	1.52	0.27	0.56	0.06	0.48
Quantofix N-volumeter	104	-0.05	0.05	0.98	0.02	0.95
Agros N meter	104	0.23	0.09	0.99	0.05	0.81
Conductivity meter	103	0.20	0.06	0.89	0.03	0.89

Separation, dilution and washing of cattle slurry from the herbage have been used to reduce ammonia volatilisation and improve dry matter yields compared to whole slurry. NH_3 volatilisation from whole slurry could be reduced by 50% by separating through a 0.3 mm sieve or by diluting with 87% water or washing with

53% water and was shown to increase herbage dry matter yield by 23% (Frost, 1990). However, separation of slurry through fine mesh size is difficult to achieve in practice and dilution and washing of slurry require costly transportation costs for water.

When applying slurry in the field, an equal rate and uniform distribution is required to prevent excesses and deficiencies. Band spreaders and trailing shoe application results in more uniform distribution of slurry to the soil surface compared to splash plate spreading (coefficient of variation of 7%) (Huijsmans and Hendriks, 1992).

The N_2O emissions from NH_4^+-N based fertilisers or slurry are relatively low when applied to grassland. However, the timing of slurry application relative to the application of chemical fertiliser N can have a large effect on N_2O emission. Significant N_2O emissions have been shown whenever cattle slurry and NO_3^--N fertiliser were applied together or within a few days of each other (Stevens & Laughlin, 2001). In a follow up study Stevens & Laughlin (2002) showed that when cattle slurry was applied 3 to 4 days prior to fertiliser NO_3^--N the N_2O emission was negligible. They suggested that the enhanced N_2O production was due to rapid oxidation of volatile fatty acids, formed during storage of slurry, stimulating denitrification, and that these were metabolised within 4 days.

Anaerobic digestion (AD) of slurry converts organic matter to biogas methane in the absence of oxygen. The digestion of the slurry in this way increased pH and NH_4^+-N by increased mineralisation of organic N (Pain *et al.*, 1990). Digestion lowers the concentration of volatile fatty acids in slurry (Paul and Beauchamp, 1989) which are a source of available carbon for denitrifiers, hence denitrification should be less for digested slurry than undigested slurry. Stevens *et al.* (1995) compared anaerobically digested slurry and undigested slurry in a laboratory experiment, and showed that overall treatments the recovery of NO_3^--N from digested and undigested slurry was 98% and 50% respectively, after 7 days. The removal of volatile fatty acids by digestion prior to slurry application could be an effective mitigation strategy for N_2O emission. Across Europe there are large numbers of on-farm digesters. In Germany there are approximately 4000 digesters and in Denmark approximately 20 digesters. At AFBI Hillsborough an anaerobic digester was commissioned in March 2008, to digest slurry from 300 dairy cows. Although the primary aim of the anaerobic digester was to produce methane to be utilised as an energy source, it may also be valuable as a means of reducing N_2O emissions when cattle slurry is applied on grassland. Key issues are the high initial capital cost and the need for local information on potential methane recovery rates from AD of animal manures produced by local livestock systems. A major research programme is currently underway within AFBI to address these issues and to determine whether AD is a potential mitigation strategy for N_2O emissions from slurry.

Modification of N Fertilisers

A *'Slow- or controlled-release fertiliser'* is defined as a fertiliser containing a plant nutrient in a form which delays its availability for plant uptake and use after application, or which extends its availability to the plant significantly longer than a reference "rapidly available nutrient fertiliser" such as ammonium nitrate, urea, ammonium phosphate or potassium chloride. Such delay of initial availability or extended time of continued availability may occur by a variety of mechanisms. These include controlled water solubility of the material (by semi-permeable coatings, occlusion, or by inherent water insolubility of polymers, natural nitrogenous organics, protein materials, or other chemical forms), by slow hydrolysis of water-soluble low molecular weight compounds, or by other unknown means.

Although slow-release fertilisers deliver the nutrient at a slower rate than conventional fertilisers, the pattern, rate and duration of nutrient release are not easily controlled as they may be affected by both handling and soil conditions (Shaviv, 2005). However, the characteristics of nutrient release for controlled-release fertilisers are more predictable.

Slow and controlled-release fertilisers have been shown to reduce N leaching (Hanafi *et al.*, 2002) and gaseous emissions (Di *et al.*, 2007). However, their use in agriculture is limited, despite recent technological developments. Only about 10% of total production is used on agricultural crops. The remainder is used for non-agricultural markets (e.g. lawns, golf courses, fruit trees and vegetables) (Shaviv, 2005). The main reason for their limited use seems to be the high cost, which may be 4-10 times higher than the cost of conventional fertilisers.

A *'Stabilised nitrogen fertiliser'* is defined as a fertiliser to which a nitrogen stabiliser has been added. A nitrogen stabiliser is a substance added to a fertiliser which extends the time the nitrogen component of the fertiliser remains in the soil in the urea or ammoniacal form. Examples of nitrogen stabilisers include:

- *Urease inhibitor:* A substance which inhibits hydrolytic action of the urease enzyme on urea. When applied to soils the effect of the urease inhibitor is to lower ammonia volatilisation from urea.

- *Nitrification inhibitor:* A substance that inhibits the biological oxidation of ammoniacal nitrogen to nitrate nitrogen. Nitrification is a key process in soil having implications for the environment as non mobile NH_4^+ is converted to mobile NO_3^-, which is the form of N most prone to loss.

Stabilised N fertilisers (urease and nitrification inhibitors) have received considerable attention recently due to their potential to lower N losses to the environment and increase crop yields. However, the efficacy of urease and nitrification inhibitors can be quite variable depending on the crop, soil properties, climatic and management factors. Urease inhibitors delay the rate of urea hydrolysis to NH_4^+-N and hence prevent localised zones of high pH and NH_4^+-N concentrations in soil, which are conducive to NH_3 volatilisation. They are expected to be most beneficial on soils where NH_3 volatilisation from urea fertiliser is high. The urease inhibitor N- (n-butyl) thiophosphoric triamide (trade name Agrotain®) is the most successful commercially. It has been shown to lower NH_3 volatilisation from surface-applied urea and increase yields on a wide range of crops. AFBI has demonstrated the efficacy of Agrotain for grass production in Ireland (Watson, 2000; Watson & Miller, 1996; Watson *et al*., 1998b, 2008). In recent agronomic trials with grassland and tillage land (DEFRA, 2006), Agrotain lowered NH_3 emission from urea by an average of 70% (range 41-100%) in 31 experimental measurements. Generally, Agrotain amended urea will increase crop yield compared to unamended urea if environmental conditions are conducive to high NH_3 loss. Agrotain is now licensed or sold in over 70 countries worldwide, in both agricultural and amenity markets (Watson *et al.*, 2009).

Nitrification inhibitors slow down the rate of oxidation of NH_4^+-N to NO_3^--N and hence reduce NO_3^--N leaching and the production of NO and N_2O by both nitrification and denitrification. They are likely to have greatest benefit on soils where N losses due to leaching or denitrification are large. The most extensively studied commercially available nitrification inhibitors are nitrapyrin, dicyandiamide (DCD) and 3,4-dimethylpyrazole phosphate (DMPP) (Weiske *et al.*, 2001; Di and Cameron, 2004). DCD is the most widely used nitrification inhibitor in New Zealand because it is cheap, non volatile (unlike nitrapyrin) and soluble in water. Although the % reduction in NO_3^--N leaching and N_2O emissions in New Zealand are relatively large, the amounts of conserved N in the soil due to DCD tend to be small (Monaghan *et al*., 2009; Smith *et al*., 2008b). The use of nitrification inhibitors for agricultural crops is still limited, due to their relatively high cost. For them to be more widely accepted for use in agriculture, they will need to be priced competitively. As their benefits include increased crop production, improved crop quality, greater management flexibility and reduced environmental losses, there should be increased interest in their use, particularly if carbon credits for lower greenhouse gas emissions can be used to offset costs. Current work is ongoing to evaluate the efficacy of nitrification inhibitors in Ireland.

Conclusions

Nitrogen is used very inefficiently in agricultural systems leading to high losses by ammonia volatilisation, during nitrification and by leaching, erosion, surface runoff and denitrification. The loss of N represents an economic loss to farmers and has environmental implications for water quality and greenhouse gas emissions. Good management strategies to improve N use efficiency are based on matching N supply to crop demand and include fertiliser type, amount and timing of application and manure management with respect to timing and method of spreading. Combining a range of management strategies will have a significant impact on the flow and excesses of N in agricultural systems. Urease and nitrification inhibitors are potentially useful tools for reducing gaseous emissions and NO_3^--N leaching. However, their effectiveness depends on the crop, soil properties, climatic and management factors. Current work is ongoing to evaluate the cost effectiveness of nitrification inhibitors in Ireland.

References

Bussink, D.W., Huijsmans, J.F.M. & Ketelaars, J.J.M.H. (1994). Ammonia volatilisation from nitric-acid-treated cattle slurry surface applied to grassland. *Netherland Journal of Agricultural Science* 42, 293-309.

Carton, O.T. & Magette, W.L. (1999). Land spreading of animal manures, farm wastes and non-agricultural organic wastes - Part 1: Manure (and other organic wastes) management guidelines for intensive agricultural enterprises. Project 4026. Teagasc, Dublin, Ireland.

Coulter, B.S. (2004) Nutrient and trace management advice for grassland, tillage, vegetable and fruit crops. Teagasc, Wexford, Ireland.

Crutzen, P. J. (1981). Atmospheric chemical processes of the oxides of nitrogen, including nitrous oxide. In: Delwiche, C.C. (ed.). *Denitrification, nitrification and atmospheric nitrous oxide*. pp. 17-44. John Wiley & Sons, New York. ISBN 9780471048961

Department of Agriculture and Rural Development, (2009). Northern Ireland Fertiliser Statistics Available at: www.dardni.gov.uk/index/publications/pubs-dard-statistics/fertiliser2009.htm (last accessed 29 January 2010).

Department for Environment, Food and Rural Affairs (DEFRA). (2006) Report for DEFRA Project NT2605 (CSA 6579). Work package 1b Ammonia emissions and crop N use efficiency (2005); Work package 2 The effect of N fertiliser forms on nitrous oxide emissions (2006); Work package 3 Optimum use of nBTPT (Agrotain) urease inhibitor (2005); Work package 4 Nitrogen losses to surface and groundwaters (2006); Work package 5 Assessment of crop phytotoxicities (2006). Available at: http://randd.defra.gov.uk (last accessed 05 Nov 2009).

Di, H.J. & Cameron, K.C. (2004). Treating grazed pasture soil with a nitrification inhibitor, Eco-n™, to decrease nitrate leaching in a deep sandy soil under spray irrigation - a lysimeter study. *New Zealand Journal of Agricultural Research* 47, 351-361.

Di, H.J., Cameron, K.C. & Sherlock, R.R. (2007). Comparison of the effectiveness of a nitrification inhibitor, dicyandiamide, in reducing nitrous oxide emissions in four different soils under different climatic and management conditions. *Soil Use and Management* 23, 1-9.

European Union (EU) (2000). European Parliament establishing a framework of the Community action in the field of water policy (EU Water Framework Directive). *Directive 2000/60/EC*.

European Union (EU) (2009). European Parliament amending Directive 2003/87/EC so as to improve and extend the greenhouse gas emission allowance trading scheme of the Community. *Directive 2009/29/EC.*

Frost, J.P. (1990). Effect of separation, dilution and washing of cattle slurry on ammonia volatilisation and grass yield. *Irish Journal of Agricultural Research* 29, 84-86.

Frost, J.P., Gilkinson, S.R. & Binnie, R.C. (2007). Methods of spreading slurry to improve N efficiency on grassland. *Proceedings of the BGS/BES/BSAS Conference, Keele University,* Staffordshire, 17-19 April 2007, pp 82-87.

Hanafi, M.M., Eltaib, S.M., Ahmad, M.B. & Syed Omar, S.R. (2002). Evaluation of controlled-release compound fertilisers in soil. *Communications in Soil Science and Plant Analysis* 33, 1139-1156.

Huijsmans, J.F.M. & Hendriks, J.G.L. (1992). Slurry distribution of spreaders and injectors. *Proceedings of Agricultural Engineering Conference*, Swedish Institute of Agricultural Engineering, Uppsala, Sweden, 1-4 June, pp 622-623.

Hyde, B. & Carton, O.T. (2005). Manure management facilities on farms and their relevance to efficient nutrient use. *Proceedings of Fertiliser Association of Ireland.* Winter Scientific Meeting. UCD, Dublin, pp 27-43. Fertiliser Association of Ireland, Dublin, Ireland.

IPCC (2007). *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.* Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. and Miller, H.L. (eds.). 966pp. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Jarvis, S.C., Wilkins, R.J. & Pain, B.F. (1996). Opportunities for reducing the environmental impact of dairy farming managements: a systems approach. *Grass and Forage Science* 51, 21-31.

Klarenbeck, J.V., Pain, B.F., Phillips, V.R. & Lockyer, D.R. (1993). A comparison of methods for use in the measurement of ammonia volatilisation following the application of livestock wastes to land. *International Journal of Environmental Analytical Chemistry* 53, 205-218.

Korevaar, H. & den Boer, D.J. (1990). Practical measures to reduce nutrient losses from grassland systems. Proceedings No. 301. pp 34. The Fertiliser Society, London.

Lockyer, D.R., Pain, B.F. & Klarenbeek, J.V. (1989). Ammonia emissions from cattle pig and poultry wastes applied to pasture. *Environmental Pollution* 56, 19-30.

Ministry of Agriculture, Fisheries and Food (2000). Fertiliser recommendations for agricultural and horticultural crops (RB209). The Stationery Office, Norwich, U.K. ISBN 0-11-243058-9

Moal, J.F., Martinez, J., Guiziou, F., & Coste, C.M. (1995). Ammonia volatilisation following surface applied pig and cattle slurry in France. *Journal of Agricultural Science* 125, 245-252.

Monaghan, R.M., Smith, L.C. & Ledgard, S.F. (2009). The effectiveness of a granular formulation of dicyandiamide (DCD) in limiting nitrate leaching from a grazed dairy pasture. *New Zealand Journal of Agricultural Research* 52, 145-159.

Pain B.F., Misselbrook, T.H., Clarkson, C.R. & Rees, Y.J. (1990). Odour and ammonia emissions following the spreading of anaerobically-digested pig slurry on grassland. *Biological Wastes* 34, 259-267.

Pain, B.F. & Thompson, B.B. (1989). Ammonia volatilisation from livestock slurries applied to land. In: Hansen, J.A. and Henriksen, K. (eds). *Nitrogen in organic wastes applied to soils.* pp 202-212. Academic Press, London.

Paul, J.W. & Beauchamp, E. G. (1989). Relationship between volatile fatty acids, total ammonia, and pH in manure slurries. *Biological Wastes* 29, 313-318.

Pretty, J.N., Brett, C., Gee, D., Hine, R.E., Mason, C.F., Morison, J.I.L., Raven, H., Rayment, M.D. & van der Bijl, G. (2000). An assessment of the total external costs of UK agriculture. *Agricultural Systems* 65, 113-136.

Shaviv, A. (2005). *Controlled release fertilizers.* In: IFA International Workshop on Enhanced-Efficiency Fertilisers. June 28-30. Frankfurt, Germany. International Fertilizer Association, Paris.

Smith, K.A., Chambers, B.J. & Williams, J.R. (2008a). Towards sustainable recycling of farm manures; in search of practical solutions to technical problems. In: Matsunaka, T. & Sawamoto, T. (eds.). *Animal Manure – Pollutant or Resource?* pp 13-39. Rakuna Gakuen University Extension Center, Ebetsu, Hokkaido, Japan.

Smith, L.C., de Klein, C.A.M. & Catto, W.D. (2008b). Effect of dicyandiamide applied in a granular form on nitrous oxide emissions from a grazed dairy pasture in Southland, New Zealand. *New Zealand Journal of Agricultural Research* 51, 387-396.

Stevens, R.J. & Laughlin, R. J. (2002). Cattle slurry applied before fertiliser nitrate lowers nitrous oxide and dinitrogen emissions. *Soil Science Society of America Journal* 66, 647-652.

Stevens, R.J. & Laughlin, R.J. (2001). Cattle slurry affects nitrous oxide and dinitrogen emissions from fertiliser nitrate. *Soil Science Society of America Journal* 65, 1307-1314.

Stevens, R.J. & Laughlin, R.J. (1997). The impact of cattle slurries and their management on ammonia and nitrous oxide emissions from grassland. In: Jarvis, S.C. & Pain, S.C. (eds.) *Gaseous nitrogen emissions from grasslands*. pp 233-256. CAB International, Oxon, UK.

Stevens, R.J., Laughlin, R.J. and Malone, J.P. (1998). Measuring the mole fraction and source of nitrous oxide in the field. *Soil Biology and Biochemistry* 30, 541-543.

Stevens, R.J., O'Bric, C.J. & Carton O.T. (1995). Estimating nutrient content of animal slurries using electrical conductivity. *Journal of Agricultural Science (Cambridge)* 125, 233-238.

Tunney, H. (1984). Slurry-meter for estimating dry matter and nutrient content of slurry. In: Williams, J.H., Guidi, G. & Hermite, P.L. (eds.). *Long-term Effects of Sewage Sludge and Farm Slurries Applications* pp 216-233. Elsevier Applied Science, London.

United Nations Framework Convention of Climate Change (UNFCCC). Greenhouse gas emission data available at: http://unfccc.int/ghg_data/items/3800.php

van der Hoek, K.W. (1998). Nitrogen efficiency in global animal production. *Environmental Pollution* 102, S1, 127-132.

van Kessel, J.S. & Reeves III, J.B. (2000). On-farm quick tests for estimating nitrogen in dairy manure. *Journal of Dairy Science* 83, 1837-1844.

Watson, C.J. (2000). Urease activity and inhibition – principles and practice. Proceedings No 454. 39pp. The International Fertiliser Society, York, UK. ISBN 085310 090 X

Watson, C.J. & Miller, H. (1996). Short-term effects of urea amended with the urease inhibitor N-(n-butyl) thiophosphoric triamide on perennial ryegrass. *Plant and Soil* 184, 33-45.

Watson, C.J., Akhonzada, N.A., Hamilton, J.T.G. & Matthews, D.I. (2008). Rate and mode of application of the urease inhibitor N-(n-butyl) thiophosphoric triamide on ammonia volatilization from surface-applied urea. *Soil Use and Management* 24, 246-253.

Watson, C.J., Jordan, C., Lennox, S.D., Smith, R.V. & Steen, R.W.J. (2000). Inorganic nitrogen in drainage water from grazed grassland in Northern Ireland. *Journal of Environmental Quality* 29, 225-232.

Watson, C.J., Laughlin, R.J & McGeough, K.L. (2009). Modification of nitrogen fertilisers using inhibitors: Opportunities and potentials for improving nitrogen use efficiency. Proceedings No 658. 40pp. The International Fertiliser Society, York, UK. ISBN 978-0-85310-295-3.

Watson, C.J., Poland, P. & Allen, M.B.D. (1998b). The efficacy of repeated applications of the urease inhibitor N-(n-butyl) thiophosphoric triamide for improving the efficiency of urea fertilizer utilization on temperate grassland. *Grass and Forage Science* 53, 137-145.

Watson, C.J., Stevens, R.J., Garrett, M.K. & McMurray, C.H. (1990). Efficiency and future potential of urea for temperate grassland. *Fertilizer Research* 26, 341-357.

Watson, C.J., Stevens, R.J., Steen, R.W.J., Jordan, C. & Lennox, S.D. (1998a). *Minimizing nitrogen losses from grazed grassland*. Agricultural Research Institute of Northern Ireland 71st Annual Report, p.49-61.

Weiske, A., Benckiser, G. & Ottow, J.C.G. (2001). Effect of the new nitrification inhibitor DMPP in comparison to DCD on nitrous oxide (N_2O) emission and methane (CH_4) oxidation during 3 years of repeated applications in field experiments. *Nutrient Cycling in Agroecosystems* 60, 57-64.

Whitehead, D.C. (1995). *Grassland Nitrogen*. CAB International, Wallingford, UK.