Nitrates Directive Consultation Document
The evidence base for assessing the impacts of the NVZ Action Programme on water quality across England and Wales.

Report by: David Johnson, Robin Hodgkinson, Eunice Lord, Martyn Silgram, Bruce Cottrill, Richard Gooday, Katrina Morrow, Sarah Smith and Adele Hulin

Checked by: Adrian Collins and Peter Dampney

Date: December 2011

Submitted to:
<client name, address if available>

Prepared by:
ADAS UK Ltd
Woodthorne
Wergs Road
Wolverhampton
WV6 8TQ

Tel: 01902 754190
Fax: 01902 743602

CONTENTS

The evidence base for assessing the impacts of the NVZ Action Programme on water quality across England and Wales

1 INTRODUCTION

1.1 Scope of report

This document presents a brief overview of the key evidence underpinning the development of the NVZ Action Programme (AP) for England and Wales. Topics covered include the current status of water quality, the requirements of the Nitrates Directive, the impact of agricultural practice on water quality, and current farming practice. It is intended that this report is part of the evidence base for the 'Consultation on Implementation of the Nitrates Directive in England'. Annex 1 of this report reproduces the questions from the consultation and indicates where relevant evidence can be found in this document.

The document focuses particularly on measurement data from commercial farming situations within NVZs, as monitored as part of Defra's NIT18 project (WT0749NVZ), to ensure that the information is as relevant and realistic as possible.

A full science report on the NIT18 option 2 project (WQ0100NIT) which measured the export of phosphorus, sediment and non-nitrate forms of N from the NIT18 micro-catchments, will be delivered at the end of that specific project in winter 2011-12. This will include detailed commentaries on the monitoring data and descriptions of the development and refinement work undertaken on the APT model (ADAS Pollution and Transport) to make it suitable for improved assessment of NVZ AP measures. A final science report on the core NIT18 project, which measures nitrate leaching within micro- catchments in surface water NVZs and from fields in groundwater areas, will be delivered in summer 2012 when this specific project is completed.

2 SOURCES OF EVIDENCE

2.1 The NIT18 project: Evaluation of the NVZ Action Programme

The NIT18 project was commissioned in 2004 to assess the impact of the NVZ AP as required under the Nitrates Directive. The project involves monitoring soils, crops and waters on a representative range of commercial farms within NVZs, evaluation of these data and other evidence, model development, synthesis to assess impacts of the NVZ AP, and assistance to Defra with reporting to the European Commission (EC).

The measurement programme is intended to reflect reality within existing NVZs. Monitoring locations, Figure 2.1 have been selected to represent a mixture of real farm practice, climate and soil/land use combinations, Figure 2.2. Farm characteristics are compared with national surveys to ensure that the study areas are representative.

Monitoring is undertaken at field scale so that the link between cause and effect can be clearly demonstrated. This allows us to investigate the impact of land use and real farm practice on water quality. Within groundwater micro-catchments, all data are for single fields. Within drained (surface-water) micro-catchments, monitoring of drainage is either for single fields or groups of fields comprising the micro-catchment. In the latter case, soil mineral N (SMN) is still measured on each individual field. Farm practice data is collected on a field by field basis.

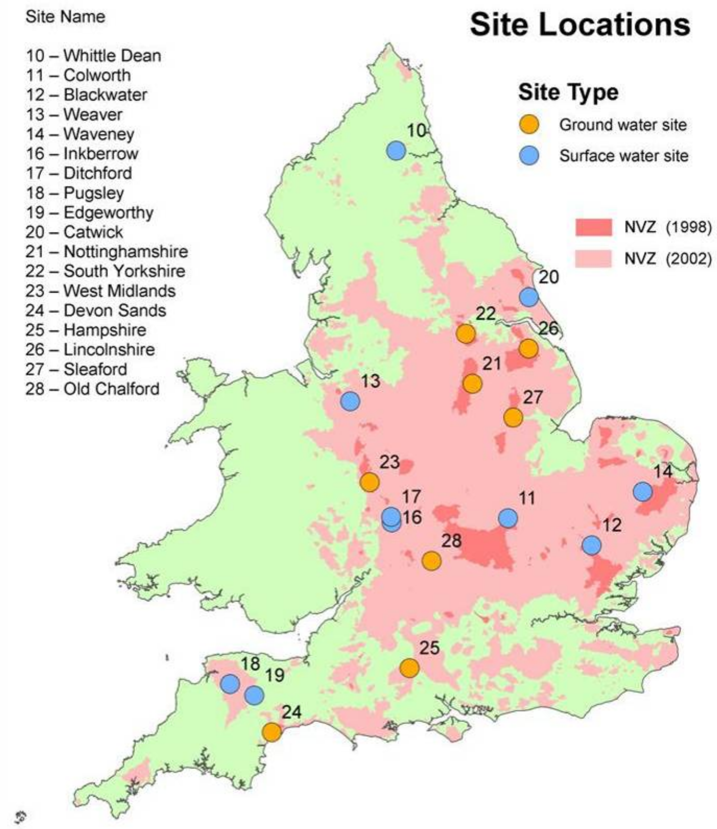


Figure 2.1. Locations of the monitored NVZ micro-catchments used in the NIT18 project.

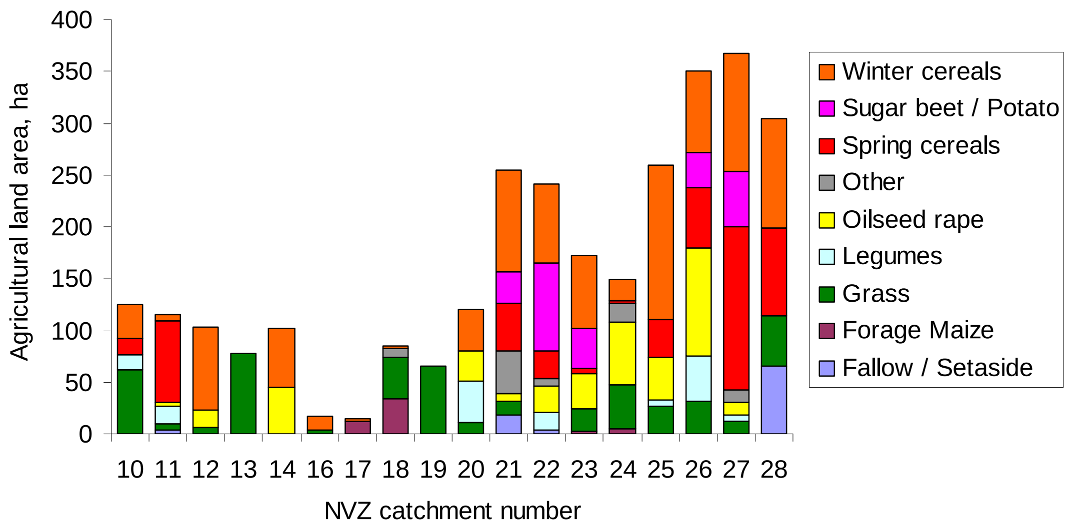


Figure 2.2. Land use within the monitored NVZ micro-catchments used in the NIT18 project in 2009. (catchments 10-20 are surface water micro-catchments dominated by clay soils; catchments 21-28 are groundwater micro-catchments with generally lighter soils, some of which were previously monitored under the NSA scheme)

From 1994, monitoring of NSAs (Nitrate Sensitive Areas) focused on groundwater micro-catchments (with permeable soils). This was extended in 2004 to include clay soils in surface water dominated micro-catchments. This extension to the monitoring programme allowed the project to improve the evidence base for surface water dominated micro-catchments (which now constitute a significant proportion of the NVZ area) and areas where pollutant transfer to surface waters responds differently to management practices. It also satisfied the need for more evidence on the potentially contrasting impacts of measures such as manure management on nitrate versus other pollutants from clay soils.

Additional monitoring of other pollutants, phosphorous (P), ammonium-N (NH_4), dissolved organic nitrogen (DON) and sediment has also been carried out during the period 2008-2011 (under the NIT18 option 2 project) to gain an integrated picture of the impact of farming on the water environment. Most pollutants other than nitrate are absorbed by the soil. To reach streams, they have to travel by routes with minimal soil interaction i.e. surface runoff, or 'preferential flow' (e.g. macropore or crack flow) to drains. Monitoring a wider range of pollutants has allowed assessment of the impacts of land management, and of the NVZ AP measures, on these other pollutants, and has built up the (previously limited) evidence base for the behaviour of these pollutants in drained soils. It has also allowed the testing and improvement of multi-pollutant models which can then be used to assess the pollution swapping potential of the AP measures. The data gathered on commercial farms complements the controlled experimentation recently funded by Defra.

2.2 Other research

The assessment of the impacts of agriculture on water quality draws on a wide range of other research over many years. These sources are referenced in the text. Particularly important in this context is Defra's 'Cracking Clays' project (WQ0118) which assesses the effect of date of application of manures on pollutant losses from clay soils to water and air.

2.3 Surveys of agricultural practice

In order to assess the impacts of agriculture on water quality, we need statistics on current farming practice and changes. The following main sources of data from Defra-managed surveys have been used:

- Annual Agriculture Survey (June Survey): crop areas and livestock numbers
- British Survey of Fertiliser Practice (BSFP): fertiliser, lime and manure inputs to crops and grass, timing of manure and fertiliser inputs.
- Farm Practice Survey (FPS, e.g. Defra (2007)): this survey collects data on a wide variety of subjects, the questions changing from year to year. Information includes slurry and manure storage, stock management, sources of information used by farmers, measures to avoid water pollution, and economic pressures.
- NIT18 project: this project collects detailed management data from over 300 fields each year. It provides essential background information on farm management practices where other sources are not available.

2.4 Water quality data: groundwaters and surface waters

The quality of ground and surface waters across England, based on data collected by the Environment Agency (EA), are reported to the EC on a four yearly cycle under Article 10 of the Nitrates Directive (Report on the state of implementation of the Nitrates Directive in the United Kingdom (England)). The most recent report (2007) evaluated the measured water quality at monitoring stations across England and adjacent marine waters, for the previous and current reporting period under the Nitrates Directive. Data were taken from 2002 and 2003 inclusive, to represent the previous reporting period, and data from 2006 and 2007 inclusive, to represent the current reporting period. The Article 10 report provides a comprehensive, presentation of all data, regardless of the reasons for collection.

The underpinning data used is from the EA's General Quality Network (GQA) for Surface Water, and the National Groundwater Quality Monitoring (NGWQM) network for Groundwater. However, the methodology required for the analysis of monitoring data for Article 10 reporting is rather simplistic and could be misleading if used in isolation. To minimise the likelihood of being mislead by this data, where possible, the following sections are augmented by more recent (unpublished) data analysis from the EA. The latest assessment of trends in surface water and groundwater nitrate data will be available in the autumn of 2011.

Groundwater (GW). The main findings from the 2007 Article 10 report were that average GW concentrations of nitrate were below 50mg/l in 83% of the monitoring network, with peak concentration below 50mg/l in 75% of the network. Trends in GW quality are difficult to assess with confidence and interpret meaningfully due to the complex nature of hydrogeology. However, in 2007, 70% of monitoring points were assessed as having stable trends with the remaining points evenly split between statistically significant upward and downward trends. In the 2008 NVZ designation, which used far more sophisticated statistical approaches, a very similar picture emerged.

Interpretation of this GW quality data at a national scale can be misleading as there are a number of confounding factors that need to be taken into account, including:

- The long unsaturated zone flow, and saturated zone residence times in many UK aquifers (section 6.2);
- Aquifer vulnerability which plays a significant role in determining groundwater quality. Thick low permeability glacial till deposits reduce infiltration and any associated pollution;
- The potential for de-nitrification in confined groundwater systems;
- The reliance on abstraction boreholes for groundwater quality monitoring data which can be closed down or moved for operational reasons in response to changing water demand or quality;
- Localised pollution which is not representative of catchment scale impacts.

These factors taken together mean that for every national scale conclusion about the current state of, and trends in, GW nitrate concentrations, there will be significant contradictory monitoring evidence. Figure 2.3 demonstrates some of these contradictions. The only way to resolve this apparently contradictory evidence is to develop a detailed local conceptual understanding of the aquifer in question. Without this local understanding, the policy message from the monitoring data can easily be undermined by data from individual, or groups of, alternative monitoring points. The archive of porous pot soil leachate measurements from the NIT18 project provides a powerful supplementary dataset which allows clearer policy messages to be defined.

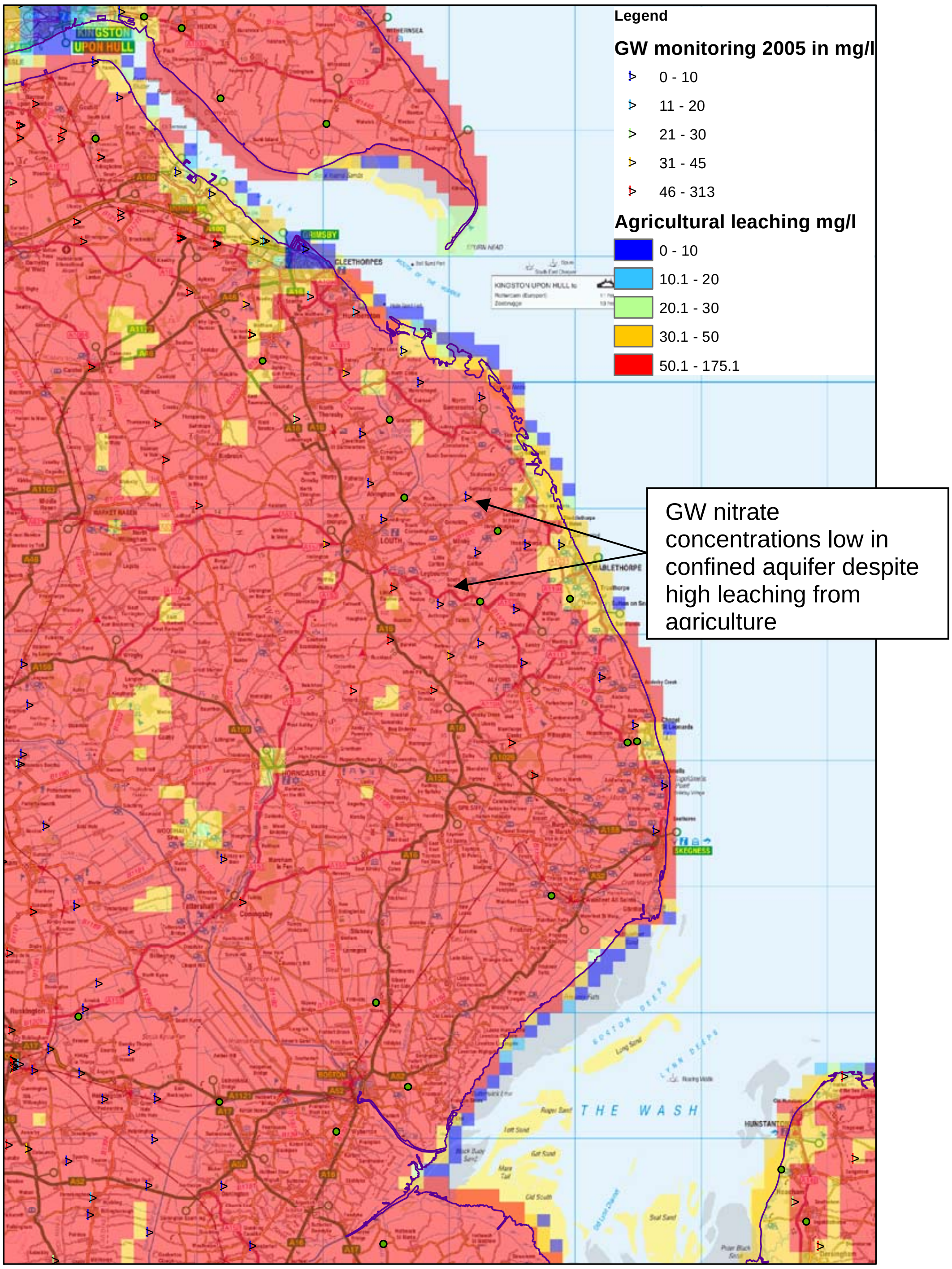


Figure 2.3. Map of GW nitrate concentrations for the 2008 NVZ review vs. nitrate leaching. (monitoring data from the Environment Agency GWQMN)

Although the water quality dataset for nitrate is by far the most detailed dataset available for groundwater, there is a growing database of other water quality data from the NGWQM network. This dataset, coupled with the baseline assessment of groundwater quality carried out by BGS (Edmunds *et al.,* 2003) allows a more complete understanding of the current groundwater quality, associated trends and sources of pollution (Johnson *et al.,* 2007). Due to the sparse nature of these data at this time, they are more relevant to local than national scale assessment and interpretation.

Surface Water (SW). The 2007 report to the EC under Article 10 of the Nitrates Directive showed that average surface water concentrations of nitrate at Environment Agency monitoring points were below 50 mg/l in 91% of the monitoring network, with maximum concentrations below 50 mg/l in 67% of the network. Trends in SW quality showed that 47% of monitoring points were assessed as having stable trends, with 33% having a statistically significant downward trend and 20% a strong upward trend. In the 2008 NVZ designation, which used a more sophisticated statistical approach, a similar picture emerged in terms of trends. In both cases, there were more monitoring points with falling than rising trends. Reduction in concentrations might be expected in grassland catchments on the basis that N inputs to grass and N surplus have fallen substantially in recent years (Lord and Johnson, 2011) and there is a need for detailed data analysis to relate observed trends in surface waters to possible causal factors.

The monitoring data on phosphate, while less complete than nitrate, also forms part of the Article 10 reporting to the EC. Mean phosphate concentrations in English rivers and lakes were below 2 mg/l PO_4 at 87% of the Environment Agency monitoring sites. The highest concentrations were to the east of the country near centres of population. Phosphate concentrations had fallen at 80% of the sites compared to the previous Article 10 report. This trend is corroborated by the corresponding picture from the farmgate phosphate balance which shows significant falls in phosphate surplus in grassland and arable systems with a near zero surplus in the arable sector (Lord and Johnson, 2011).

Data on pollution incidents between 2003 and 2007 showed a significant reduction of direct pollution from agriculture (typically relating to ammonium, Biological Oxygen Demand or pesticides). Between 2003 and 2007, SW incidents related to agriculture went down by 34% from 872 to 572, with serious incidents down by 38% from 98 to 61. Over the same period, water industry related incidents went down by 20%, with the most serious incidents reduced by over 50%. More recent data from 2011 shows that while incidents related to the water industry have continued to fall, agriculture related incidents have remained static (pers comm., Environment Agency). The Dairy sector is identified as the cause of over 40% of the category 1 and 2 pollution incidents associated with agriculture, compared to 12% for arable and 11% for beef.

Worrall *et al.* (2009) compared water quality data for selected catchments within and outside NVZs across England and Wales and found little evidence that existing NVZ APs have had a significant impact on nitrate concentrations or trends in surface waters.

3 IMPACTS OF AGRICULTURE ON NITRATE LOSSES.

3.1 Interpretation of the data

Wherever possible, the information below is given as (mean) nitrate concentrations in leachate (mg/l NO_3). These averages are flow-weighted (i.e. they give the concentration as if all leachate and water for the whole winter had been gathered into a bucket and the resulting concentration of the mixture was measured). In some cases, nitrate losses are given as kg N per ha.

In some cases, late autumn SMN data are reported. These refer to Soil Mineral Nitrogen (SMN, nitrate-N plus ammonium-N) within the top 90 cm of soil depth. SMN is often measured on commercial farms in spring as part of assessing the soil N supply (SNS) for crops. Measurements in autumn give an indication of relative nitrate leaching risk. High SMN is correlated with high nitrate loss (Figures 3.1 and 3.2). However the relationship varies according to faming system and climate because:

- crop uptake in autumn reduces N leaching (e.g. grassland has higher SMN but lower losses than arable cropping where crops are often small or absent in autumn);
- additional mineralisation may occur during autumn e.g. after break crops;
- nitrate concentrations are greater in drier years or locations, but losses are greater in wetter years/locations and on free-draining soil types.

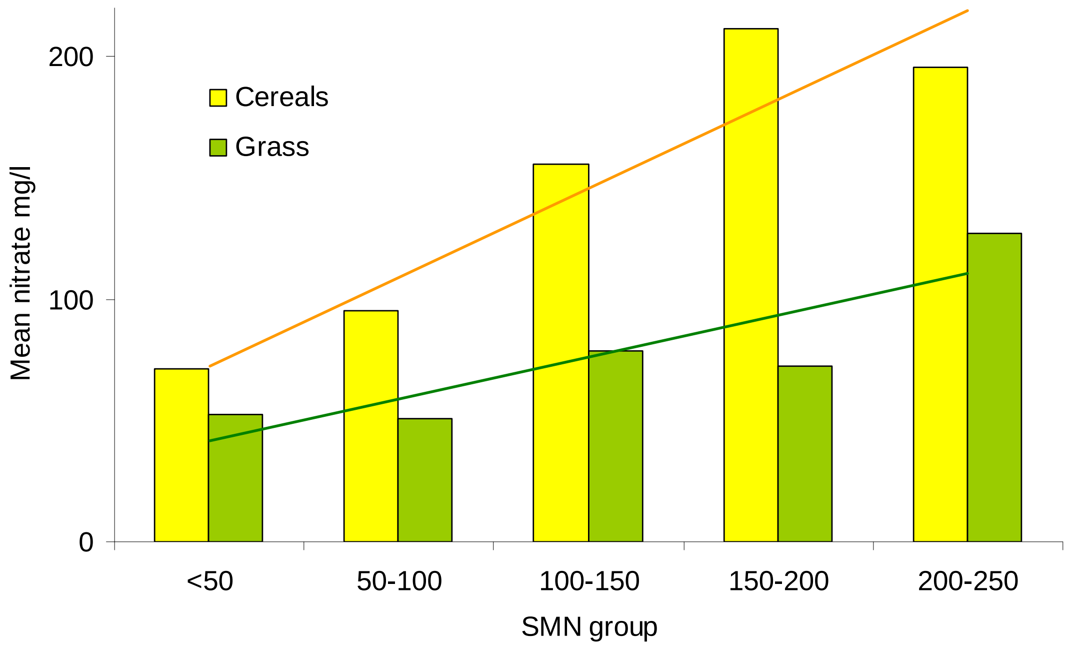


Figure 3.1. Nitrate concentration in leachate is correlated with the amount of nitrogen in the soil in late autumn (SMN, kg N/ha in 0-90 cm). The relationship differs for grass vs. cereals cropping (linear regression lines shown). Source: NIT18 project.

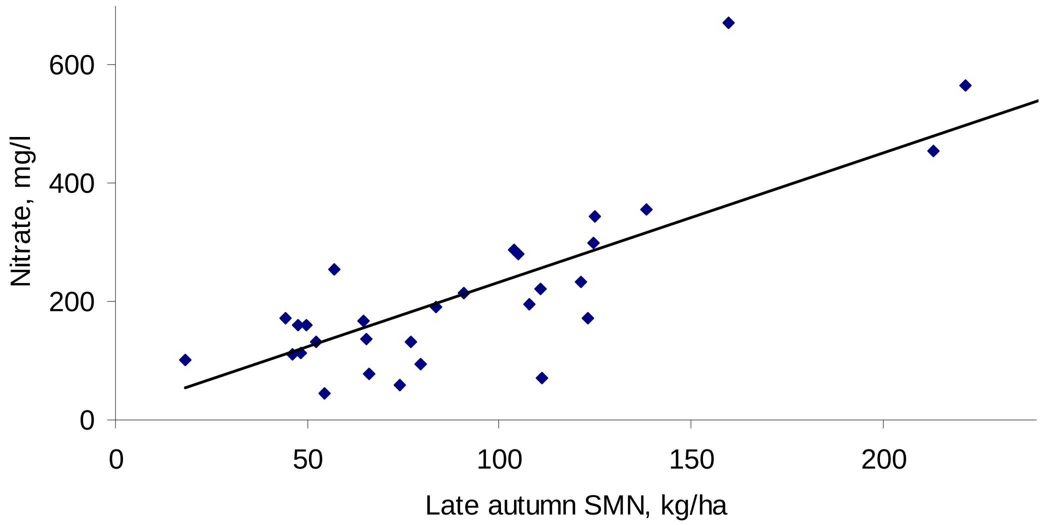


Figure 3.2. High SMN following potato crops is associated with high nitrate concentrations in leachate. Source: NIT18 project.

3.2 Typical nitrate concentrations in leachate from agricultural land

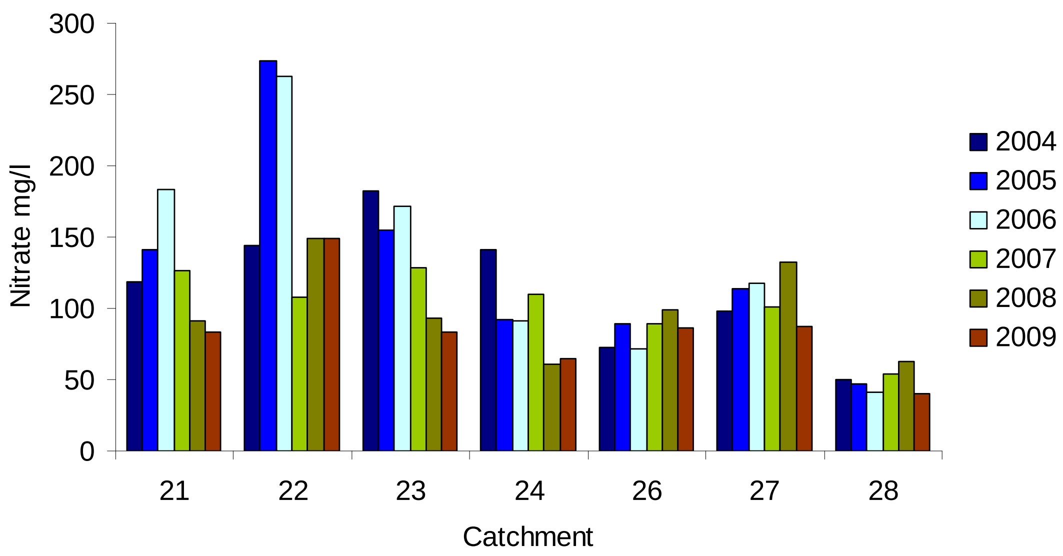


Figure 3.3 Mean nitrate concentrations in leachate from agricultural land for groundwater micro-catchments 21 to 28. Most micro-catchments exceed 50 mg/l in most years. Catchment 28 contains significant areas of unfertilised grass.

Average concentrations of nitrate from arable fields typically exceed 50 mg/l even if farmers follow best practice and NVZ measures are applied. Figure 3.3 shows that mean nitrate concentrations for the groundwater catchments 20-27 (which are dominantly arable) were above 50 mg/l in most years. Since most measurements taken in the field trials conducted as part of the NIT18 study give average nitrate concentrations in leachate for a range of rainfall scenarios, soils and locations. These measurements show that all arable crops except sugar beet leach nitrate at concentrations significantly greater than 50 mg/l.

3.3 Site factors affecting nitrate concentrations in leachate

In addition to land use (above) and land management (discussed below), there are spatial and temporal factors which affect nitrate concentrations in leachate, and total nitrate loss, chiefly soil type and local rainfall.

3.3.1 Rainfall

Total nitrate losses (kg/ha) are greater in wet winters, but average concentrations are lower due to dilution. For this reason, and the fact that nitrate concentrations tend to be smaller under grass than arable systems, the greatest nitrate concentrations are recorded in Eastern areas (Figure 3.4). Quantities of nitrate leached are fairly similar in the east and west. Winter drainage in arable areas is typically in the range 50 to 300 mm, and concentrations rarely average less than 50 mg/l, as shown in Figure 3.5.

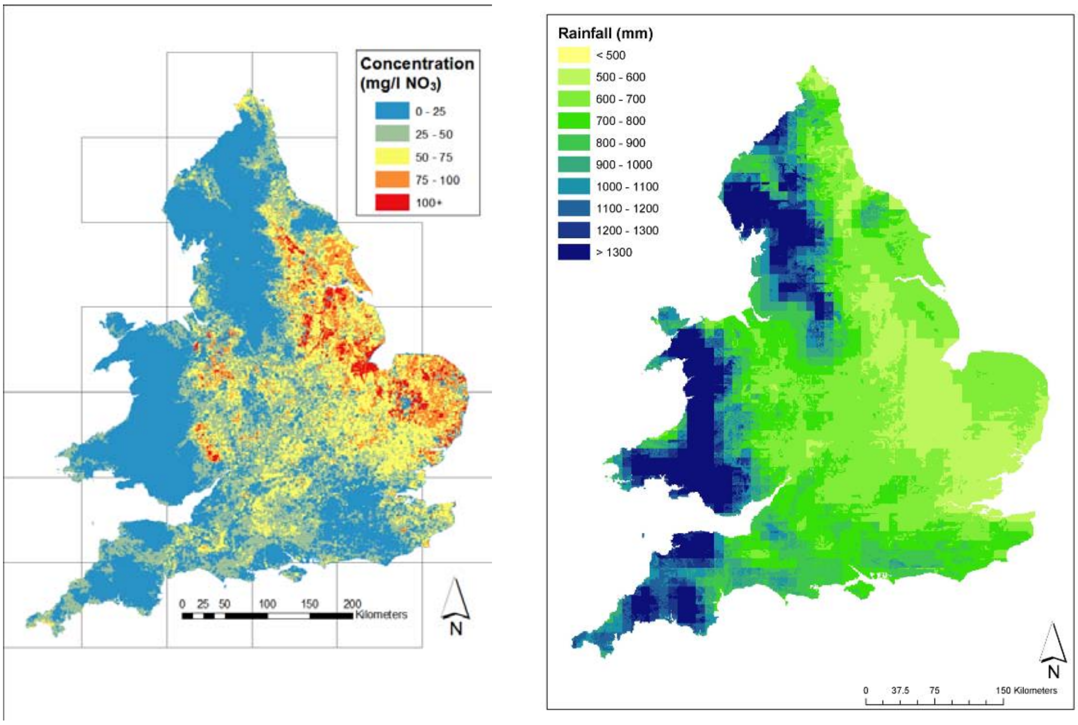


Figure 3.4. Nitrate-N concentrations are greater in eastern areas, due to lower rainfall, and a greater proportion of arable land. Source: Modelling within the NIT18 project. Note the scale of the nitrate map: 11.3 mg/l nitrate-N is equivalent to 50 mg/l nitrate.

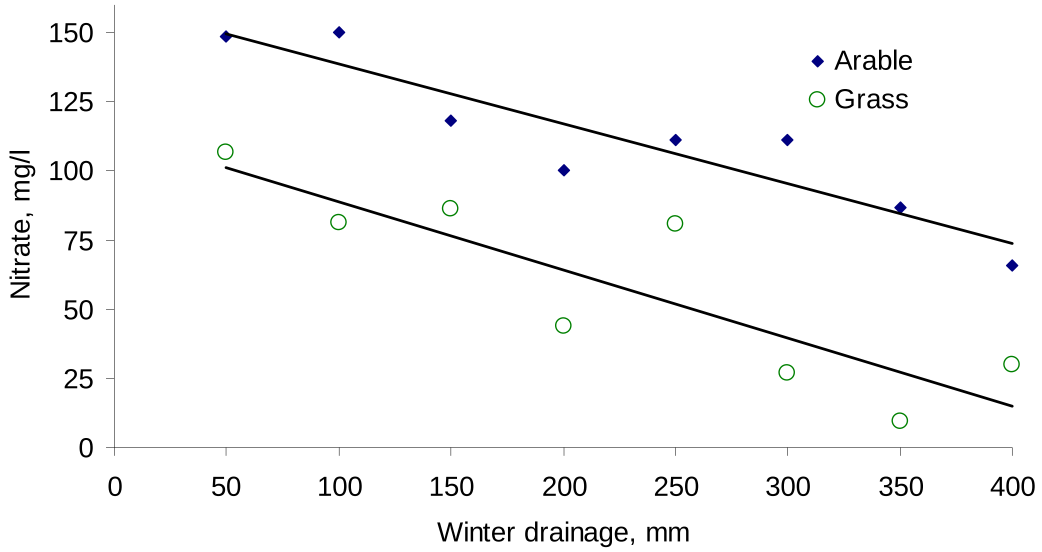


Figure 3.5. Mean nitrate concentrations from arable and grassland grouped by overwinter drainage volume. Source: NIT18 and earlier projects

3.3.2 Soils

The other main factor influencing nitrate leaching is soil type (Figure 3.6). Light sandy or shallow soils need relatively little water to leach nitrate through the soil profile. More retentive loamy, silty and clay soils need more water and may not be fully purged of nitrate under the typical rainfall conditions within most NVZs.

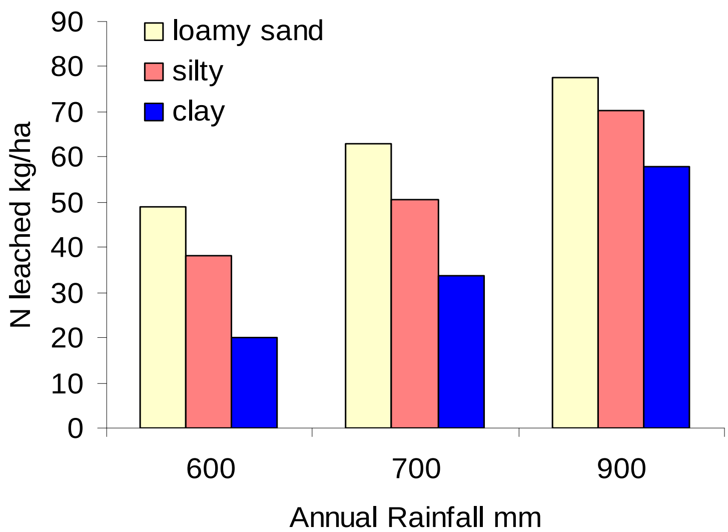


Figure 3.6. Effect of soil type and annual rainfall on nitrate leaching after winter wheat. The 'loamy sand' soil represents the lightest soils, while the 'silty' soil is more typical of permeable soils. Results using the ADAS NIPPER model.

Clay soils comprise two components of response: the slow leaching of nitrate within the clay matrix, and, during heavy rain, rapid transfer of pollutants at the soil surface via cracks to drains. Nitrate concentrations in such rapid flow are usually quite low, unless fertiliser has been recently applied. Concentrations of nitrate in drains therefore tend to fluctuate greatly from day to day in response to rainfall events, and these fluctuations are rapidly transferred to watercourses.

Mean concentrations of nitrate in drainage from clay soils may be less than those from the permeable soils found over groundwaters. However, due to dilution during heavy rains, they often exceed 50 mg/l. Furthermore, concentrations fluctuate and the upper 95th percentile concentrations (on which NVZ designations are based) will exceed the mean. On most sites monitored concentrations exceed 50 mg/l under arable cropping. For example, a site with an average nitrate concentration of only 35 mg/l (due to low N inputs and 25% non-agricultural land) might still record over 5% of winter measurements in excess of 50 mg/l nitrate. If this was one of the EA's water quality monitoring points, the upstream catchment would require designation as an NVZ. Figure 3.7 illustrates this issue with the somewhat greater nitrate concentrations more typical of arable systems.

Climate and soil type also affect land use, with arable systems concentrated in the drier areas. This, in addition to their direct impact, helps explain the systematic trend towards greater nitrate concentrations in the drier East of England, as mapped by Lord & Anthony (2000).

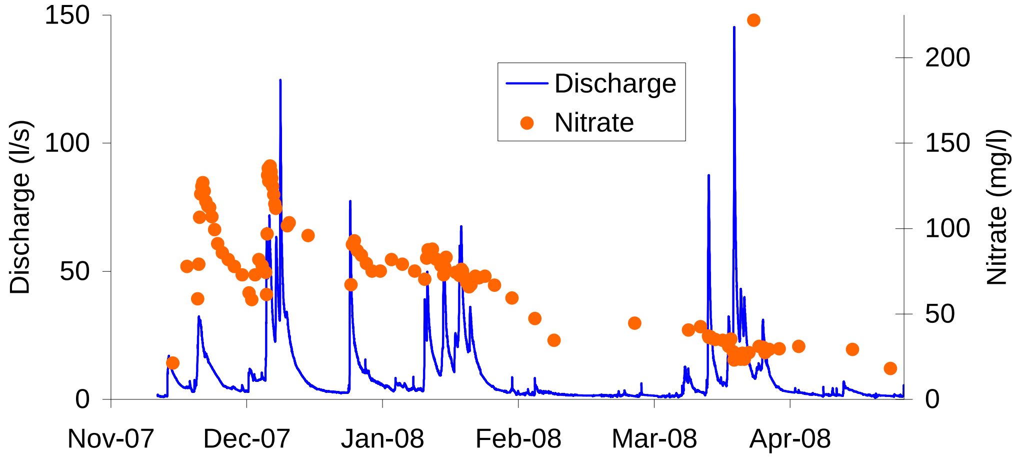


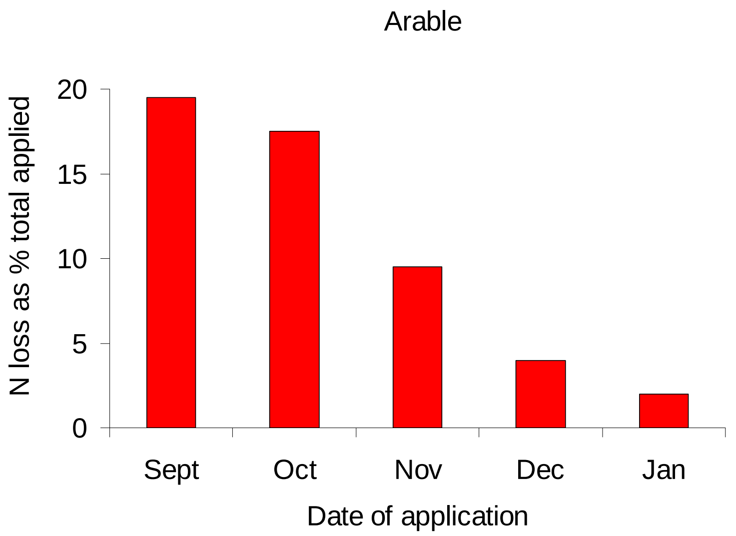
Figure 3.7. Nitrate concentrations in leachate at surface water gauge 141 in 2007/8 under arable cultivation. The flow-weighted mean nitrate concentration was 75 mg/l, and the upper 95th percentile nitrate concentration ca. 131 mg/l.

3.4 Manure application effects on nitrate loss

3.4.1 Manure timing and 'Closed Spreading Periods'

Autumn applications of slurry increase nitrate loss so that on the lightest soils and in the wetter areas, most of the readily-available N from manure will be lost by spring (Figure 3.8). The nitrate from manures applied from February onwards on light sandy soils is generally still within the root zone when spring growth starts.

Autumn-sown crop cover can reduce nitrate loss, but the soil already contains enough nitrate to supply the needs of most autumn-sown crops. Only if crops are sown very early (e.g. oilseed rape in August) will they take up the additional N supplied by autumn-applied manures. For a given application, losses are generally smaller on grassland than arable because N uptake by grass continues throughout the autumn (Figure 3.8).



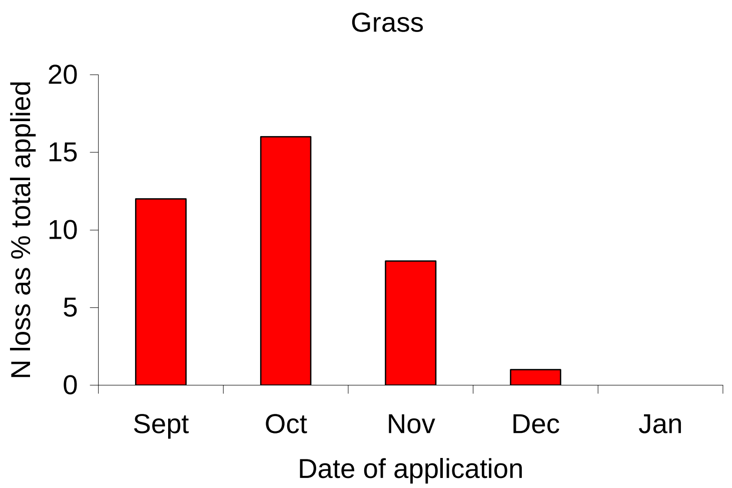


Figure 3.8. Effect of timing of slurry application on N leaching loss on sandy soils with an annual rainfall of 700-800 mm: arable v grassland. Data from porous pots. (Source: Chambers *et al*., 2000).

Dirty water from parlour washings or yard runoff often contains very high concentrations of readily available N (i.e. the proportion of nitrogen as nitrate or in forms which are rapidly converted to nitrate) which can have a significant localised impact on water quality (Figure 3.9). However, straw-based manures (farmyard manure, FYM), especially after storage, have a low content of readily available N Application of such manures in autumn has a smaller effect on nitrate leaching than an equivalent quantity of total N applied as slurry or poultry manure which have a high content of readily available N (Figure 3.10).

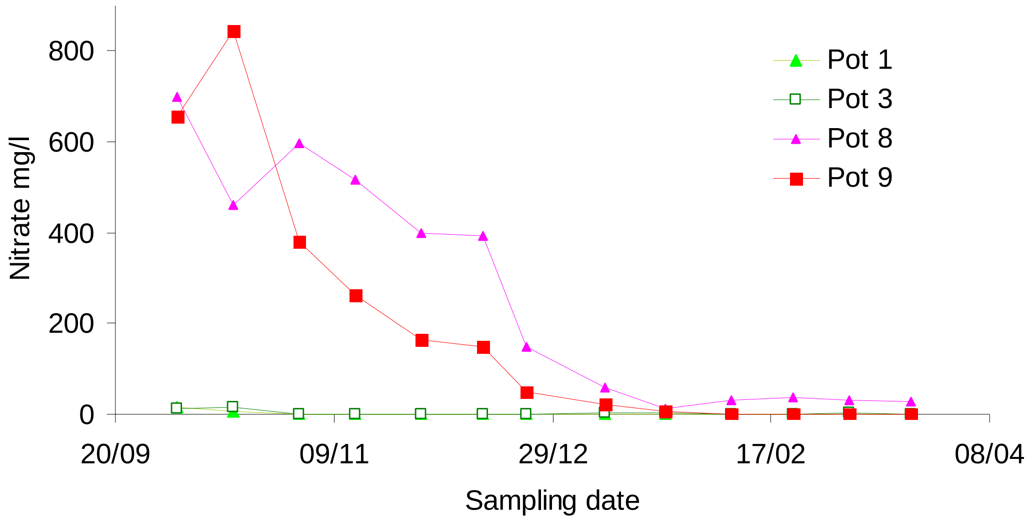


Figure 3.9. Large quantities of dairy 'dirty water' applied to grass increased nitrate concentrations (sampling points 8,9) relative to the rest of the field that received no dirty water (sampling points 1, 3). Source: NIT18 project.

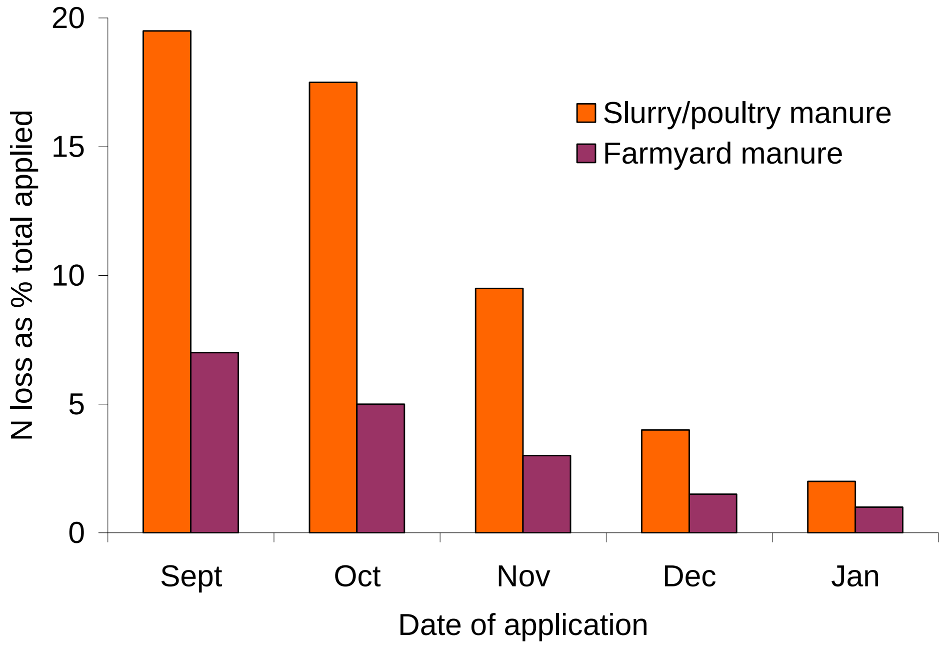


Figure 3.10. Nitrate leaching losses following livestock manure applications (Source: Chambers *et al*., 2000)

Manures contain a range of potential pollutants apart from N. Applications from January onwards have a low risk of nitrate loss, but because soils are wet, the risk of surface runoff and drain flow shortly after application is greater, especially on impermeable clay soils. Such flows risk mobilisation of other manure pollutants (e.g. phosphorus (P), ammonium (NH_4), faecal organisms) to surface water ditches and streams (but not to groundwater). The choice of timing of manure applications must take these conflicting risks into account.

Defra project WT0932 ‘Pollutant losses following organic manure applications in the month following the end of the closed period’ identified the key issues associated with a one and a two month extension to the existing closed period. The work demonstrated that nitrate leaching from cattle and pig slurry would be reduced slightly, and P loss would be reduced significantly, by extending the closed period in spring (losses associated with poultry manure would remain un-changed because most is currently applied in the autumn). However, losses of ammonia to air would generally increase because more of the manure would be applied to grass or growing crops and would therefore not be incorporated. In addition more of the manure would be applied in summer when the temperature is higher and ammonia losses increase.

3.4.2 Manures: longer term effects

Manure application, Table 3.1, increases the risk of nitrate leaching throughout the rotation, and not just for the few months after it is applied. Figure 3.11 shows how late autumn SMN remains elevated even two years after the application of manure. This is because half or more of the N in manure is in slowly-available (‘organic’) forms, which increase the soil nitrate/ammonium content over many years as the organic matter decomposes.

Table 3.1. Percentage of manure applied by month and landuse: BASELINE (data from 2007 Survey of Fertiliser Practice).

Manure type	Landuse	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	% manure to each landuse
Cattle slurry	Grassland	14	15	15	8	4	5	4	4	4	3	8	9	93
	Arable	<1	1	1	1	<1	<1	<1	1	1	1	1	<1	7
	Total	14	16	16	9	4	5	4	5	5	4	9	9	
Pig slurry	Grassland	3	6	6	3	3	4	3	5	4	0	2	2	41
	Arable	4	10	10	5	0	0	0	7	9	4	7	3	59
	Total	7	16	16	8	3	4	3	12	13	4	9	5	
Poultry	Grassland	1	3	3	1	<1	1	<1	1	0	<1	<1	<1	10
	Arable	1	9	9	5	2	2	2	18	38	1	2	1	90
	Total	2	12	12	6	2	2	2	19	38	2	2	1	

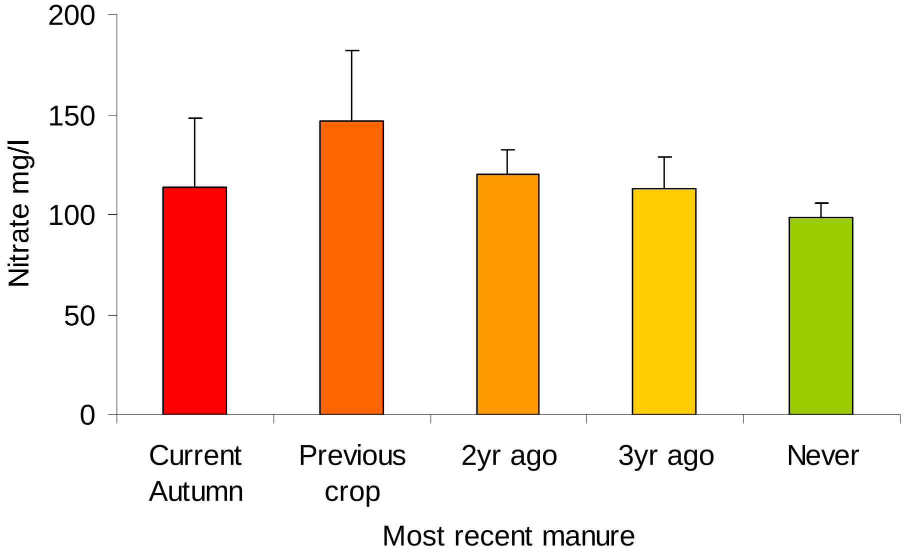


Figure 3.11. Effect of manure use on average autumn soil mineral nitrogen content following winter wheat crops. (Source: NIT18 project).

3.5 Grassland systems

In permanent grass pasture, less of the applied nitrate is leached than where arable crops are grown, because grass takes up nitrate for a longer period of the year. The nitrate leaching from grassland varies according to the intensity of the livestock system that uses the land and the nitrogen input. The most intensive systems can show losses similar to arable land.

Losses of nitrate from intensive grassland management, associated with intensive dairying, often exceeds 50 mg/l even if farmers follow best practice <u>and</u> the 2008 NVZ measures are adopted (e.g. the Livestock manure N farm limit of 170 kg N/ha). Four fields monitored during the NIT18 project showed that nitrate concentrations in excess of 150 mg/l are not unusual below intensively managed grassland. However, nitrate leaching below permanent pasture that is grazed extensively or lightly fertilised, is often well below 50 mg/l.

Losses of pollutants from grassland systems are correlated with numbers of livestock. Intensively stocked farms generate greater losses per ha and per animal kept (Jarvis, 1994). Therefore, a reduction in stock numbers reduces losses of all pollutants. This can be seen in the Figures presented by Oenema *et al. (*2009*)* for 'N flows' from agriculture. In general, the countries with the highest excretion per unit area have the lowest nitrogen utilisation efficiency (NUE) figures. This is because livestock are inefficient users of N compared to arable crops and intensive livestock rearing concentrates manures into small areas.

The greatest benefits of reducing stocking density would be felt if there was a reduction in total stock numbers. However, this is unlikely to be practicable when the NVZ area as a whole is considered.

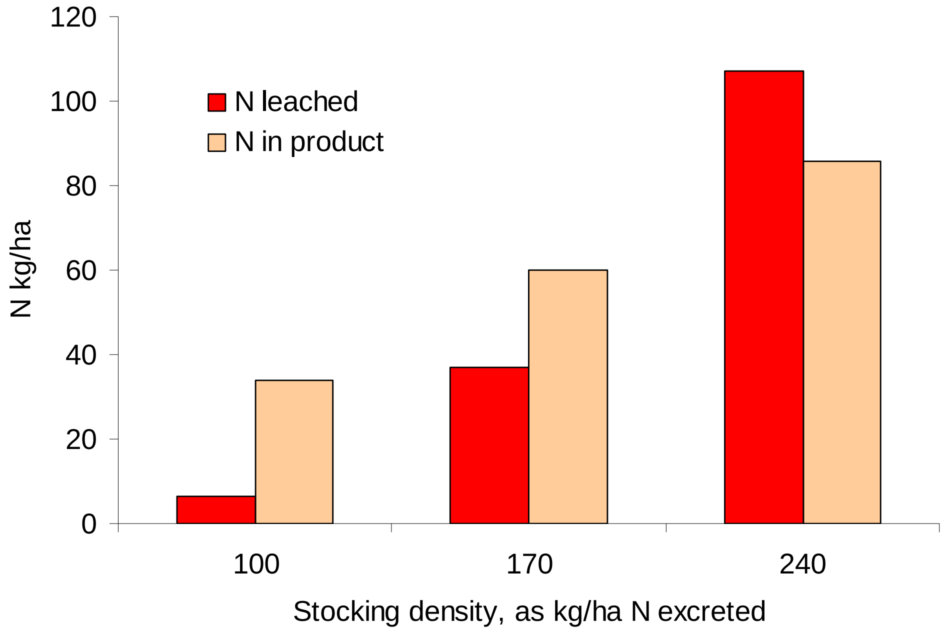


Figure 3.12. Effect of stocking density in dairy systems on nitrate leaching, and N transformed to products (milk or meat). Nitrate losses per cow, and per unit of product, increase at high stocking density in grazing systems. Source: NCYCLE model (Scholefield et al. 1991).

Leaching below permanent grass on clay soils with poor drainage is often low due to denitrification (Figure 3.21). While nitrate concentrations are reduced, there may be an increase in nitrous oxide emissions due to denitrification reactions and associated reduction in fertiliser N efficiency. The greater wetness in these situations will tend to mean increased surface runoff with associated direct pollution risks for watercourses.

3.6 Manufactured N fertiliser inputs to arable crops and closed spreading periods

3.6.1 Fertiliser inputs to arable crops

Nitrate leaching is increased at high N inputs. In arable cropping, the rate of increase is shallow at low fertiliser N inputs. However, above the economic optimum, the increase is much steeper because the crops' capacity to take up N is saturated (Figure 3.13). Additional N for crop quality (e.g. to increase the protein content of wheat for bread making) is usually used inefficiently and increases nitrate leaching.

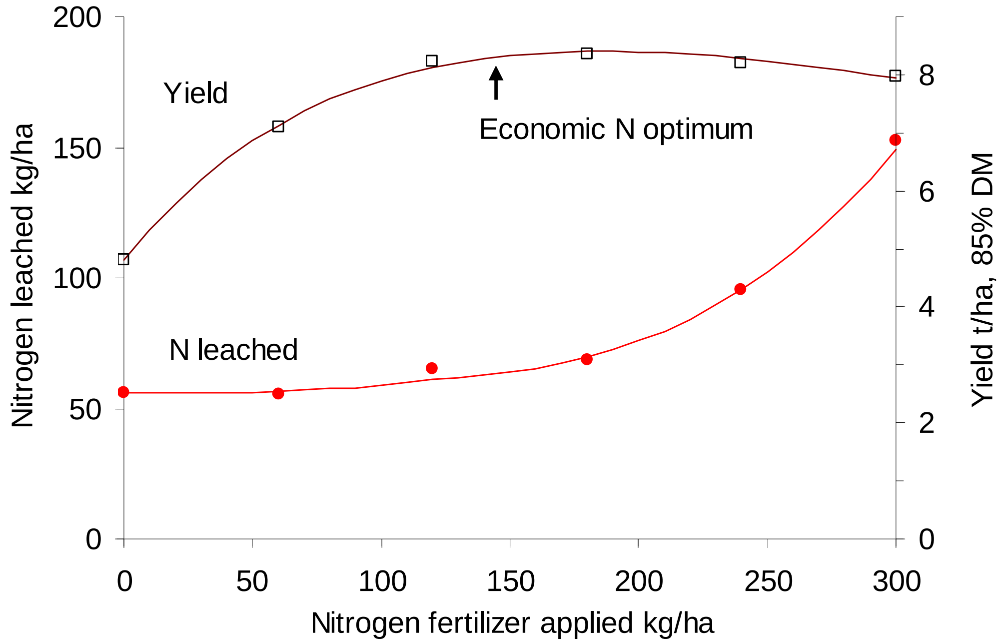


Figure 3.13. Experimental data showing relationship between fertiliser N inputs to cereals, yield, and nitrate leached over the following winter. Source: Lord & Mitchell (1998).

In grassland systems, nitrate leached is correlated with fertiliser N input, partly because this is related to stocking density (Figure 3.14). At high stocking densities, the N lost per cow is also greater – that is, the system becomes less N-efficient. However, the effects are complex and are affected by feeding regimes and manure management.

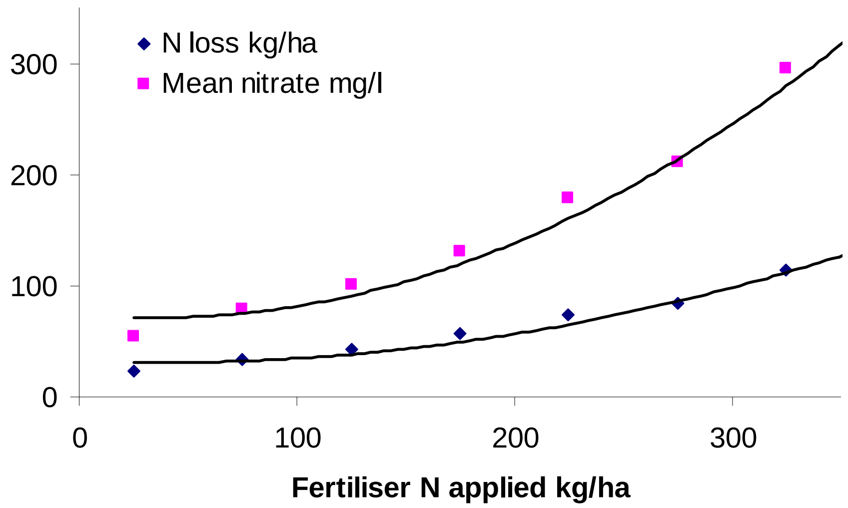


Figure 3.14. Nitrate loss from grassland as a function of fertiliser N inputs. High input sites generally were also intensively stocked. Source: NIT18 project. For Y axis see legend.

3.6.2 N fertiliser closed spreading periods

The timing of fertiliser N application also contributes to the amount of leaching. In general, autumn applications on shallow or sandy soils with high rainfall will leach below the root zone before growth starts in the spring. The only exceptions to this are crops which have a very high nutrient requirement in the autumn, most notably winter oilseed rape, which can utilise additional nitrogen in the autumn. This is recognised in the Fertiliser Manual (RB209) (Defra, 2010).

3.7 Autumn crop cover and drilling date

Nitrate losses are smaller where vigorous crop cover is present during autumn/early winter (Figure 3.15). There is a correlation between the nitrate concentrations in leachate, and the date of drilling of the autumn-sown crop.

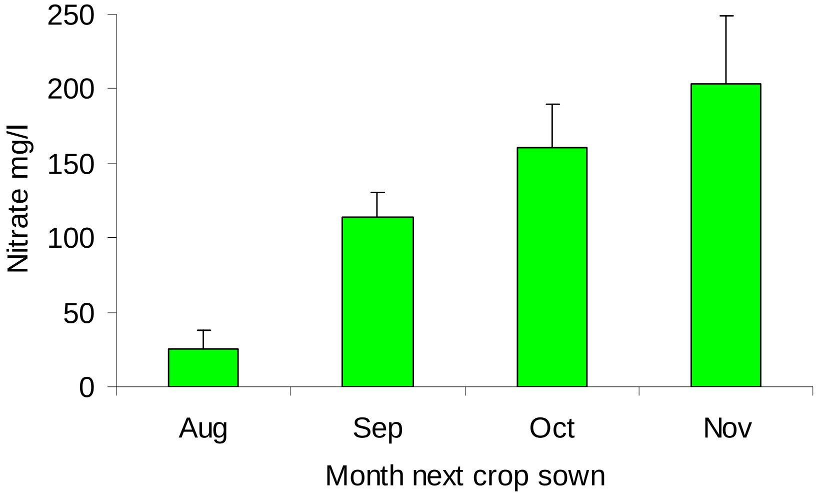


Figure 3.15. Nitrate concentrations in leachate are smallest where the next crop is established early. (SMN was similar for all sowing dates). Source: NIT18 project.

Purpose-sown over-winter cover crops have been shown to be highly effective in reducing nitrate losses (Shepherd, 1999) as shown in Figure 3.16. The approach was also proved within the Nitrate Sensitive Areas (NSA) Scheme, where it was found that early sowing and minimal soil disturbance were best to achieve the necessary rapid greening up (Lord *et al*., 1999). Destruction of weeds and volunteers was counter-productive and increased costs to farmers.

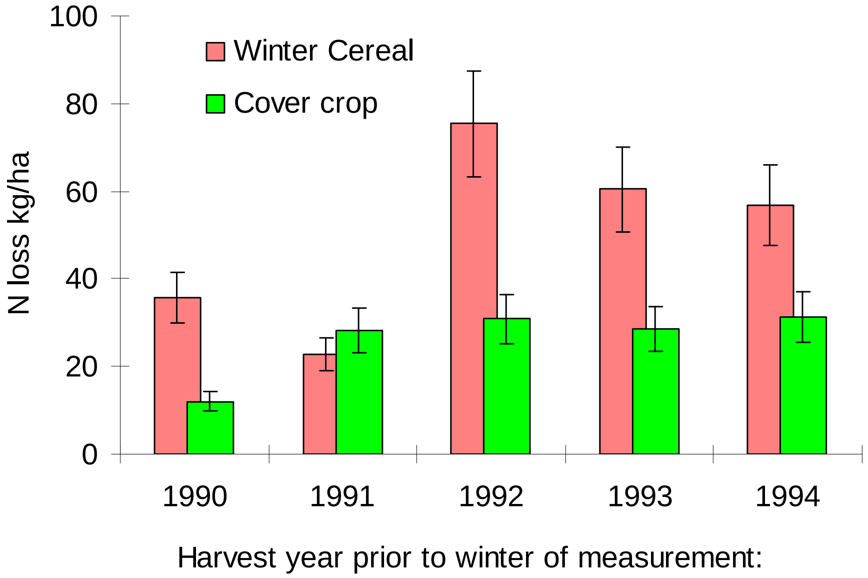


Figure 3.16. Over-winter cover crops established before spring crops were effective in reducing nitrate leaching in Nitrate Sensitive Areas. Source: Lord *et al*., 1999.

It is estimated that 30% of crops on light to medium soils, and about 20% of crops overall, are suitable for cover crops (i.e. are spring sown after a crop which is harvested before mid-September). This is about 75% of all spring-sown crops in England. The effectiveness of the cover crop decreases in dry years (Figure 3.16, 1991/92) and in drier parts of the country. This is because the nitrate is less likely to drain below the root zone by the start of the growing season in spring.

Cover crops have multiple pollutant benefits. The risk of losses of phosphorus and sediment in surface runoff is significant in autumn when the soil is bare. The provision of cover during this period reduces the risk of loss and means that significant reductions in both pollutants should be expected, compared to fields that are left bare over-winter.

Evidence for the impact of cover crops on bird numbers is patchy. It is known that over-winter stubbles have a favourable impact on farmland bird numbers. Expert opinion suggests that the ideal conservation cover crop is a stubble which has been allowed to green up naturally, typically after spring barley. Early and rapid growth of naturally occurring vegetation is possible due to the early harvest date and the fact that there is no use of pre-harvest glyphosate. The patchy nature of the resultant green cover with variable vegetation height is ideal for bird access and food supply through the 'hunger gap' in late winter. Stubble turnips have also been found to provide a positive benefit for farmland bird numbers. Cover crops which require ploughing out of stubble and which produce a uniform crop canopy, would provide little benefit for bird numbers. Early crop destruction (i.e. pre-Christmas) will provide limited food through the 'hunger gap'.

Experience gained during the NSA scheme did not suggest that cover crops represented a significant risk due to the creation of a green bridge for pests.

3.8 Crop type

Some break crops leave greater N residues compared to cereal crops, and this results in greater nitrate leaching over winter. These crops include potatoes, peas and other legumes, oilseed rape (if N inputs are high), and vegetables receiving high N inputs such as lettuce and brassicas. Nitrate leaching can also be high after fodder crops, partly because they often receive manure (Figure 3.17).

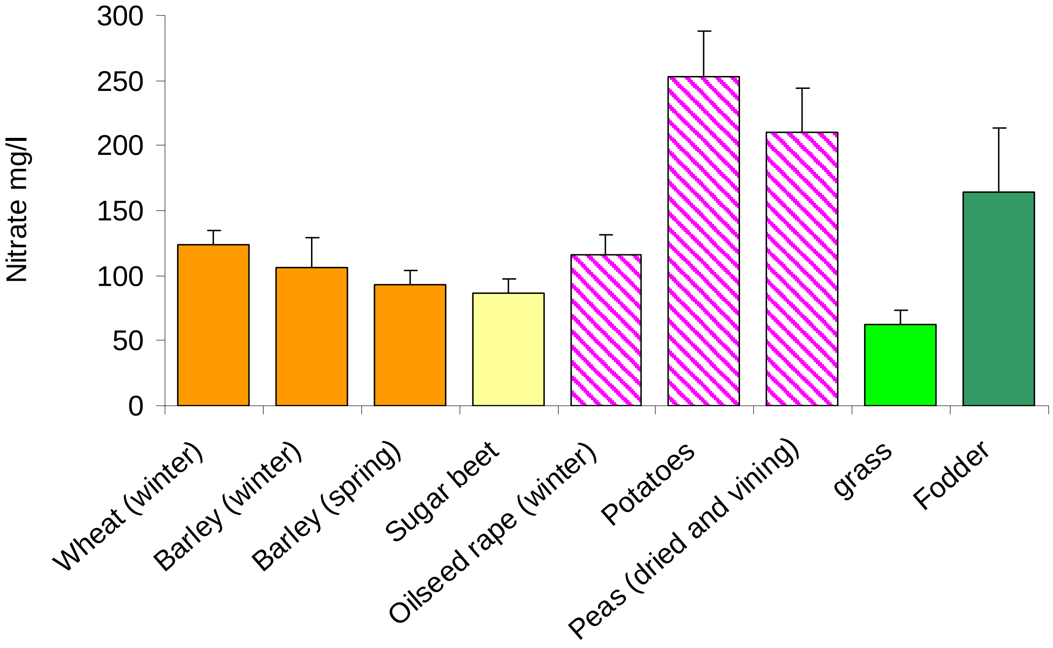


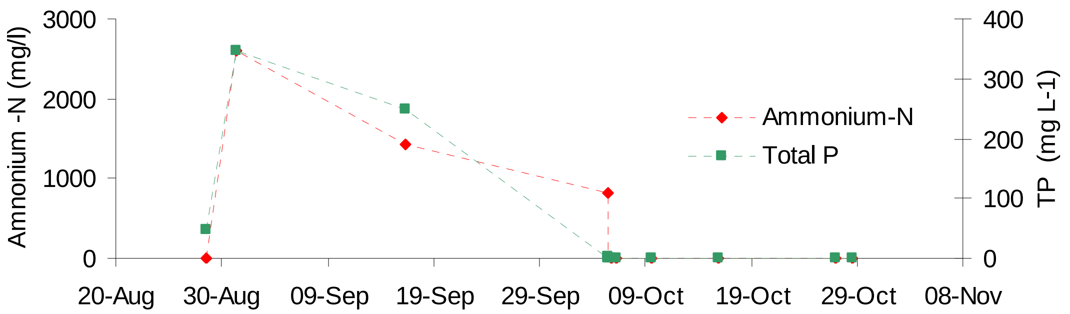
Figure 3.17. Average nitrate concentrations in winter leachate following different previous crops. Source: NIT18 project.

3.9 Field heaps of solid manure

When manure is stored in the field, nitrate and other pollutants will leach out from the heap. Long-term or repeated storage on the same site poses a risk, especially if the site is close to water. Research indicates that losses of nitrate from over-winter storage in the field are 0 to 4% of the total N in the manure (Chadwick *et al*., 2002; Sagoo *et al*., 2004). The advice to farmers in the CoGAP is only to use temporary field heaps if there is no risk of run-off polluting water. Such heaps should not be placed over field drains, nor within at least 10 m of a watercourse (ditches etc), and not within 50 m of a spring, well or borehole that supplies water for human consumption or is used in farm dairies.

If field heaps are accidentally sited over effective field drains this can result in significant pollution. The data presented in Figure 3.18a shows the concentrations of ammonium-N and phosphorus in field drain discharge when a pig FYM storage heap was sited over one of the drains (source: NIT 18 project). The volume of drain flow measured during this incident was minimal as the soil was not at field capacity and little rainfall occurred. The liquid discharging through the field drains was the concentrated "gravy" from the slurry in which the concentrations of pollutants were extremely high. In this example, where the field drain discharged into a small pond, this incident would have had a very significant environmental impact at a local scale. In contrast, the nitrate concentrations observed in leachate from the storage heap were insignificant (Figure 3.18b), although when the soil itself started to drain in early October, nitrate concentrations increased but only to a level similar to that from other field parcels within the micro-catchment. No impact of the pollutant losses from the field heap was observed at the micro-catchment scale.

(a)



(b)

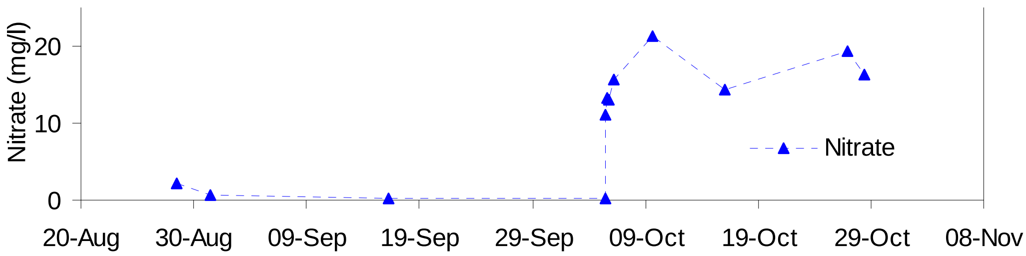


Figure 3.18. Ammonium-N, total phosphorus and nitrate N in discharge from a field drain with solid manure heap located on top of the drain. (Source NIT 18 project).

3.10 Reduced protein animal feed

There is scope for reducing nitrate leaching by reducing the protein (or N) content of livestock diets (Oenema *et al.,* 2009*)*. In doing so, it is essential to ensure that, by correct choice of feeds and supplements, diets provide the necessary protein to meet the animals' requirement. Failure to do so will result in reduced productivity and fertility, and compromised animal health and welfare.

The price of feed materials is a major factor in determining their selection as livestock feeds. As the price of high quality protein feeds increases, there is increasing pressure on 'safety margins', with the potential effect of reducing feed N content. For extensive production systems, however, there may be greater reliance on poorer quality protein feeds, resulting in more of the feed protein being excreted.

Where productivity is not compromised, reducing dietary N intake increases the efficiency of feed N utilisation. Considerable variation exists for N utilisation by livestock; for example, the average feed N utilisation efficiency for broilers is 60%, but ranges from 44% to 74% (Applegate *et al*., 2003). Much of the variation can be attributed to feed selection, livestock management, increased productivity, age of livestock, or health status. There is, therefore, considerable scope for increasing N utilisation – and thereby reducing N excretion – from the adoption of currently available technology.

The scope for further reductions in feed N levels could be increased through improved methods of feed analysis – e.g. to determine essential amino acid digestibility (for pigs and poultry) or protein degradability (for ruminants). Other options include changes in feed and livestock management, and breeding for improved productivity (Cottrill et al., 2005).

The environmental benefits of low N diets have been widely reported, and they have been adopted across many livestock sectors in the UK and EU. In the absence of any comprehensive data on feed use data in the UK, however, the extent to which they are being used here – and the scope for any further reductions in N excretion - is unknown (Lord *et al*., 2010; Clothier, 2011).

3.11 Organic farming

Registered organic farmers are not permitted to use manufactured nitrogen fertilisers, and legumes are used to fix N for use by crops and grass. Nitrate leaching from organic grassland systems is similar to that from conventional systems that receive up to about 150 kg/ha N as manufactured fertiliser. Nitrate leaching from ley-arable systems is variable. Sometimes (e.g. after ploughing out clover leys), losses are high, at other times losses are lower than from conventional systems. Production also is generally lower than from conventional systems (Stopes *et al*., 2002).

3.12 The importance of the timing of nutrient pollution for aquatic ecological impacts

Defra project WT0932 'Pollutant losses following organic manure applications in the month following the end of the closed period', identified the key issues associated with a one and a two month extension to the existing closed period. However, the impact on pollutant losses is only part of the story. Pushing slurry applications back to later in the spring caused higher peak losses of NH_4 – N ,P and E.coli as the slurry applied to wet soils entered the environment via rapid flow pathways (e.g. field drains), or overland 'runoff' flow (Defra Project WQ0118) . These 'peak' losses occur during the ecologically sensitive early to late spring, as opposed to higher overall losses in the less ecologically sensitive autumn and winter. (Edwards *et al*., 2003) (see Figure 3.19). The research also showed that pollution events were worse on grass than arable. This is almost certainly due to the greater connectivity in the soil between the soil surface and field drains, due to 'by-pass' flow in cracks/mole channels, than in annually cultivated arable soils.

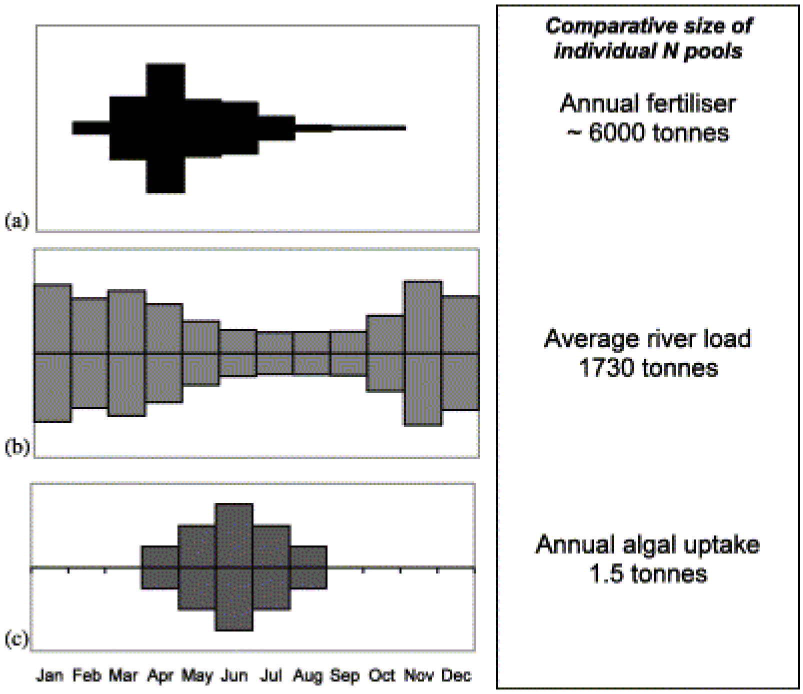


Figure 3.19. Relative timing of (a) fertiliser N applications, (b) annual river loads and (c) estimated algal N uptake, expressed on a monthly basis and for an 'average' year for the Ythan catchment in Scotland. The approximate sizes of each component are also shown for comparative purposes. From Edwards *et al*. (2003).

3.13 Surface water catchments: Mean v. 95th percentile concentrations.

Nitrate concentrations in watercourses in SW catchments are highly variable day to day. This is because during heavy rainfall, water flows rapidly to drains through cracks and larger pores, bypassing the soil matrix. This minimises contact with the pool of leachable soil nitrogen in the soil matrix, and hence bypass flow is normally characterised by low nitrate concentrations. Baseline nitrate concentrations in surface water catchments come from matrix flow, which tends to have higher concentrations due to prolonged contact with the soil. During heavy rainfall events, the influx of bypass flow dilutes the volume of matrix flow, and the overall nitrate concentration drops (Figure 3.20) (unless there has been a recent application of manure or fertiliser which can move rapidly to the watercourse as part of the bypass flow). Spikes of particularly high nitrate concentrations can occur in the spring and early summer, as shown in Figure 3.20. These have been attributed to manure or fertiliser applications. Applications in early spring carry greater risk because soils are wetter, and there is more risk of drain flow following the application.

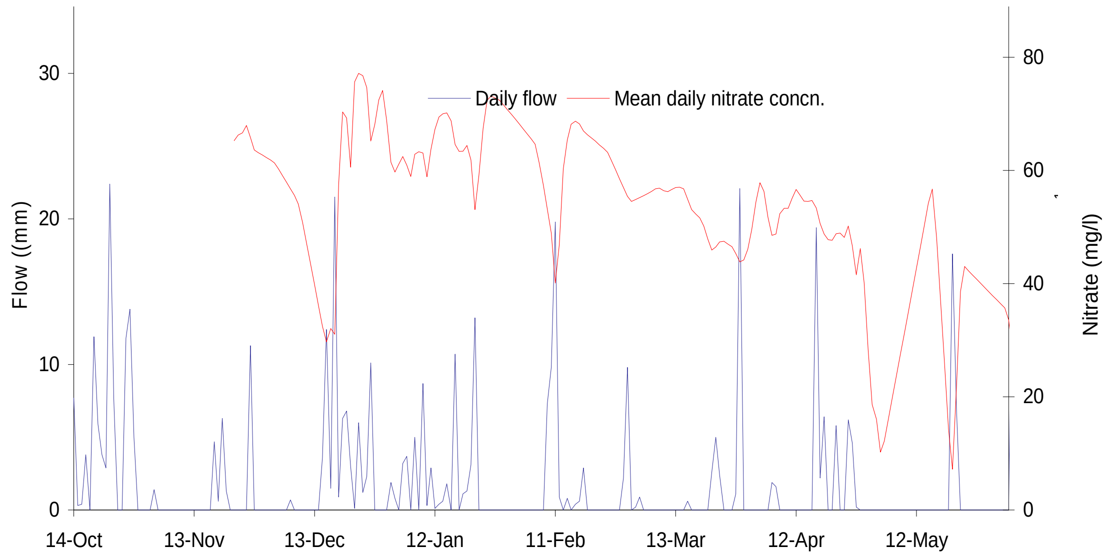


Figure 3.20. Nitrate concentrations in water draining from grassland in western England, winter 2005/6, showing dilution during heavy rain; and a gradual decline in nitrate concentrations over winter.

Grassland on heavy clay soils may leach less nitrate, especially if poorly drained, due to denitrification. Figure 3.21 shows data from two nearby clay micro-catchments, each intensively stocked with dairy cattle. Catchment 19 was poorly drained (and heavily poached) whereas catchment 18 was well-drained. Nitrate concentrations in drainage from catchment 19 were much lower than would be expected given the stocking density - this difference was attributed to denitrification as a result of soil wetness due to poor drainage. However, while catchment 19 looks acceptable from the point of view of the Nitrates Directive, conversion of nitrate to nitrous oxide by denitrification is not beneficial to the environment as it represents an increase in greenhouse gas emissions.

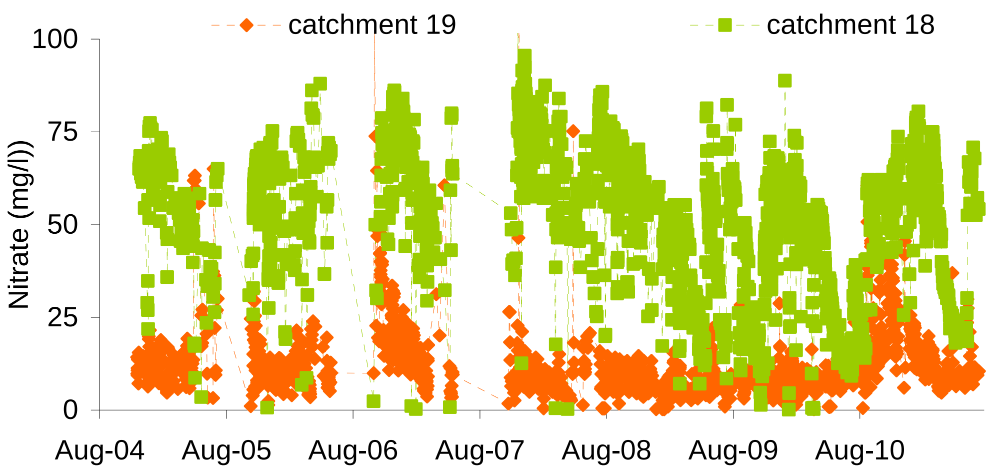


Figure 3.21. Contrasting nitrate concentrations in streams draining two nearby clay micro-catchments both supporting intensive dairying. The lower concentrations in catchment 19 were attributed to denitrification due to soil wetness. Source: NIT18 project.

Because of the great variation in nitrate concentrations from day to day, the upper 95th percentile concentration in surface water catchments is substantially greater than the mean – typically by a factor of 1.5 or more (Figure 3.22). This variation is reflected at water sampling points. Groundwaters usually show a less variable behaviour because the long travel times allow the mixing of water of different ages.

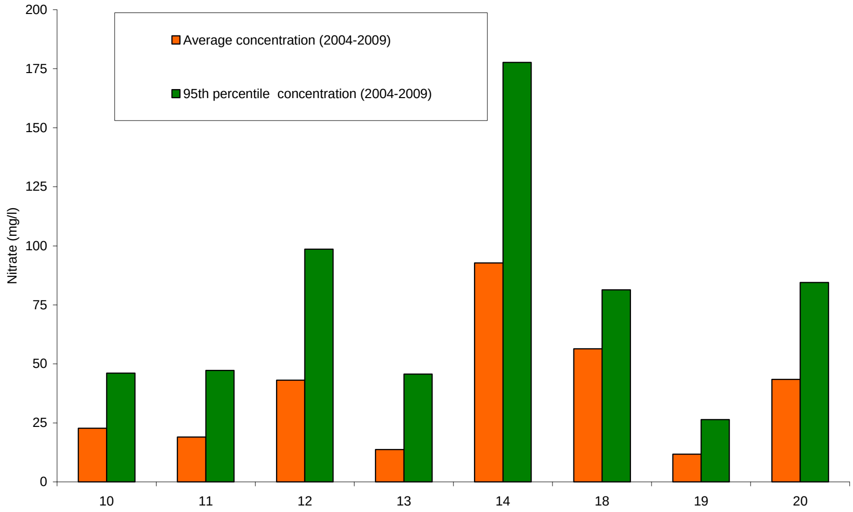


Figure 3.22. Comparison between mean and 95th percentile nitrate concentrations at surface water micro-catchments. Source: NIT18 project.

4 EFFECT OF AGRICULTURAL PRACTICES ON OTHER POLLUTANTS:

4.1 Phosphorus and ammonium loss to water

Phosphorus, ammonium, faecal contaminants and organic matter from manures can be washed into waters either in surface runoff, or in soil bypass flow to drains after heavy rain. Both of these risk pathways are greater when soils are fully wetted (i.e. in winter). In both cases, the pollutants can reach streams almost immediately. This is in contrast to nitrate, most of which reaches streams after passing through the soil profile and possibly through a groundwater system as well. This difference in behaviour is due to the fact that nitrate is not adsorbed onto soil particles, whereas the other pollutants are.

This difference in behaviour means that while applying manures from mid-winter onwards carries a relatively low risk of nitrate pollution, it carries a higher risk of direct pollution of streams with the other pollutants. Figure 4.1 shows the effect of applying slurry in early spring (March 6th) to a drained and moled heavy clay soil. Ammonium concentrations spiked sharply (and the same was observed for total dissolved phosphorus; TDP) due to rapid flow of pollutants to drains from surface-applied manure. Nitrate concentrations were little affected. Risks are greatest when heavy rain follows within a few days of application which was the case here with 13mm occurring on the 10th of March.

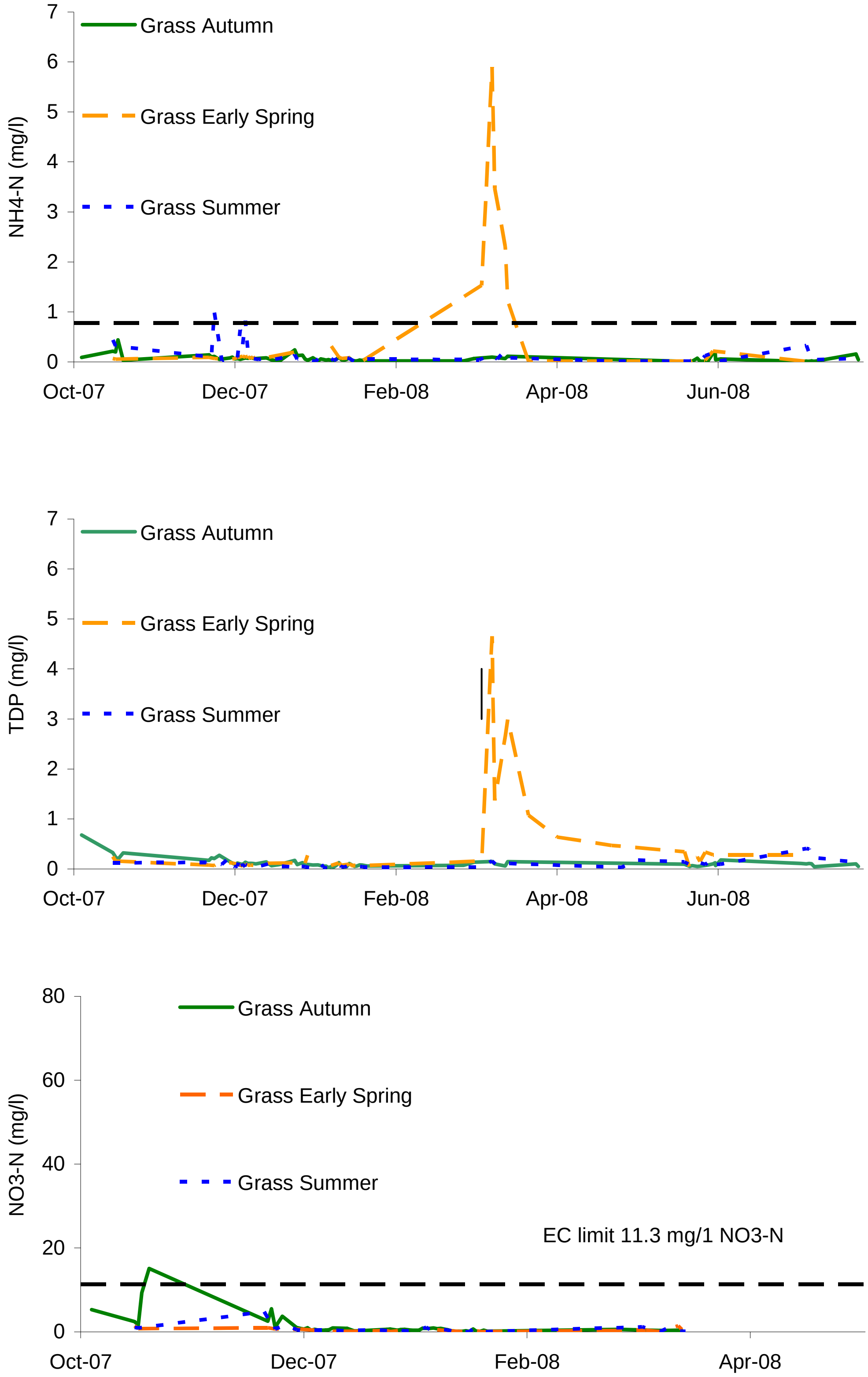


Figure 4.1. The effect of applying slurry in mid-winter to a drained and moled heavy clay soil on total dissolved phosphorus (TDP), ammonium-N and nitrate. Contrast the absence of a 'spike' in pollutant concentration following applications to drier soils (in autumn or summer). Source: Defra 'Cracking Clays.'

Dirty water also often contains high concentrations of pollutants, and the volumes applied can increase the risk of pollutant losses to watercourses. Very high concentrations of ammonium were observed in drainage from a winter wheat field on a commercial farm, following application of high volumes of dirty water to a clay soil over a prolonged period. The ammonium concentration was diluted but was still notable at the micro-catchment scale (Figure 4.2). These results are generally consistent with other work on dirty water (Williams and Nicholson, 1995) although they reported that there was a two fold reduction in concentration following passage of the dirty water through the soil to the drain. If it is assumed that some attenuation has occurred as the liquid passes through the soil, the results presented in Figure 4.2 suggest a starting concentration of in excess of 1000 mg NH_4-N per litre. Ammonium N concentrations of this magnitude are higher than would be normal for dirty water and suggests that there may have been poor management in terms of keeping yards clean, or contamination of the dirty water with slurry or 'gravy' from solid manure heaps. These concentrations observed in drain flow should be regarded as representing the extreme that is likely to be observed, rather than a typical value. More typical starting concentrations for dirty water would be in the range 150 - 500 mg NH_4-N/l, but this can still result in significant concentrations in field drain flow.

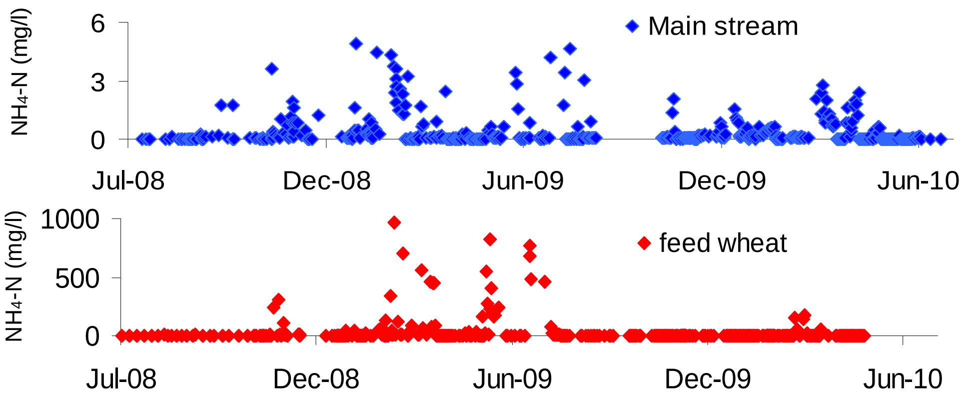


Figure 4.2. High ammonium concentrations in drains following applications of dirty water to a clay soil, and the resulting signal at micro-catchment scale. Source: NIT18 project.

4.2 Soluble organic nitrogen (SON)

Loss of N in organic compounds ('soluble organic N' or SON) can also represent a significant source of both N, and carbon for biological activity. Currently, the Environment Agency usually measures nitrate, nitrite and ammonium, and the sum of these is used to assess compliance with the Nitrates Directive. However, the EC have identified that attention should be paid to all N forms, not just nitrate, to avoid under reporting the nitrate issue. Particular attention has been focused by the EC on SON and it has been postulated that this could form a substantial part of the total N loss to water, especially in livestock areas. SON has been linked to eutrophication risk.

The need for further information on both dissolved and organic-N was identified during the review of riverine eutrophication undertaken by the Centre for Ecology and Hydrology (CEH) as part of the initial phase of project NIT18. There is evidence that soluble forms of organic N are lost, especially from grassland systems, in substantial quantities. However more quantitative data are needed to provide a basis for catchment-scale estimation of loads and mitigation impacts. Organic N is often associated with livestock excreta, as are forms of soluble P, and ammonium-N. Murphy *et al*. (1999a) reviewed the occurrence and behaviour of SON in agricultural soils. Jones *et al*., (2004) reported that SON was a significant component of the pool of soluble N in many soils. Significant leaching of SON has been reported by several authors including Siemens and Kaupenjohann (2002). The abundance, movement and transformation of SON is clearly an important area with respect to water quality but one in which there are gaps in our knowledge. For example, the importance of dissolved organic N at the catchment scale, where a wider range of loss pathways operate, remains uncertain This source of N may also have significant implications for greenhouse gases, since it will eventually denitrify and may also encourage denitrification by providing a source of C.

The importance of losses of nitrogen as SON has been investigated in all of the NIT18 micro-catchments. Mean and maximum SON concentrations over the study period to date (July 2008-June 2010) are presented in Table 4.1. The SON concentrations have been calculated by difference i.e. SON = total dissolved nitrogen – (ammonium-N + nitrate-N). We have assumed in this calculation (in accordance with evidence from river monitoring data) that concentrations of nitrite-N in surface waters are insignificant as it is a transient species. When the SON concentrations calculated by difference are small, confidence in the calculated value is reduced because the errors associated with each determinant used to derive it will dominate. Values where SON is less than 2.5% of the total dissolved nitrogen concentration have been excluded from the calculation of the mean. The largest mean SON concentration was measured at the outlet of catchment 13. This is a grassland catchment dominated by intensive dairy farming. The largest peak concentrations were measured in catchment 14 which is dominated by arable farming, but receives livestock manures from adjacent livestock units. The smallest average concentrations of SON were measured from catchments 10 (which contains a considerable proportion of extensive grassland), catchment 11 (predominantly arable with minimal manure inputs) and catchment 19 (intensive grass, extensive grass and woodland.) These data indicate some association between elevated concentrations of organic nitrogen in waters, and livestock manure applied to land.

Figure 4.4 shows a comparison between the SON concentration and ammonium-N concentration at each of the catchments across the monitoring period (1st July 2008 to 30 June 2010). The data show that at most of the sites, nitrate-N concentrations were much larger than SON, the exceptions being catchments 13 and 19 where nitrate concentrations were very low. SON concentrations were temporally invariant compared to nitrate-N; this is generally consistent with the findings of Homewood (2005) who reported that concentrations of SON in the River Test and its estuary, showed no apparent relationship with season. Analysis of the data from all of the catchments identified that the relationship between the proportion of the dissolved N load (ammonium + nitrate + soluble organic N + nitrate) (measured as SON) and nitrate-N concentrations was inverse (Figure 4.3). This suggests that the causal factors generating nitrate-N and SON in water are different, as clearly the behaviour of the two species is not coupled. For the catchments where dairy farming is practised (13, 18 and 19) higher SON concentrations are normally associated with higher ammonium-N concentrations (Figure 4.4). This suggests that the principle source of the SON in these catchments is freshly applied farm manures, for which elevated ammonium-N concentrations are a good marker. No clear relationship could be found between SON and ammonium-N for the other catchments - this suggests that freshly applied manures may be less important as a source of SON. This may be because catchments 13, 18 and 19 were predominantly in grass or maize where manures remain on the surface, in contrast to the arable systems where the manures are ploughed down and incorporated - the former situation increases the likelihood of rapid transfer of manure N into aquatic systems.

Catchment number	10	11	12	13	14	18	19	20
Mean SON (mg N/l)	1.1	1.2	2.3	4.6	2.6	2.1	1.5	2.0
Maxima (mg N/l)	3.8	5.9	10	8.4	15.1	9.5	4.2	5.0

Table 4.1 Mean Soluble Organic Nitrogen (SON) concentrations at catchment outlets (2008-2010 provisional data).

The largest mean SON concentration was measured at the outlet of catchment 13. This is a grassland catchment dominated by intensive dairy farming. In contrast, the largest peak concentration was measured in catchment 14 which is dominated by arable farming into which farm manures are imported from adjacent livestock units. Catchment 12 (predominantly arable but with a pig unit from which dirty water is spread within the catchment) produced the next largest mean SON concentration followed by catchment 18 (dairy farm and significant areas of land cropped to maize). Average SON concentrations from catchments 12 and 18 were similar to that from catchment 14. The smallest average concentrations of SON were measured from catchments10 (mainly extensive grassland), catchment 11 (predominantly arable with minimal manure inputs) and catchment 19 (dominated by a mixture of woodland and poorly utilised grass but with some dairy cows).

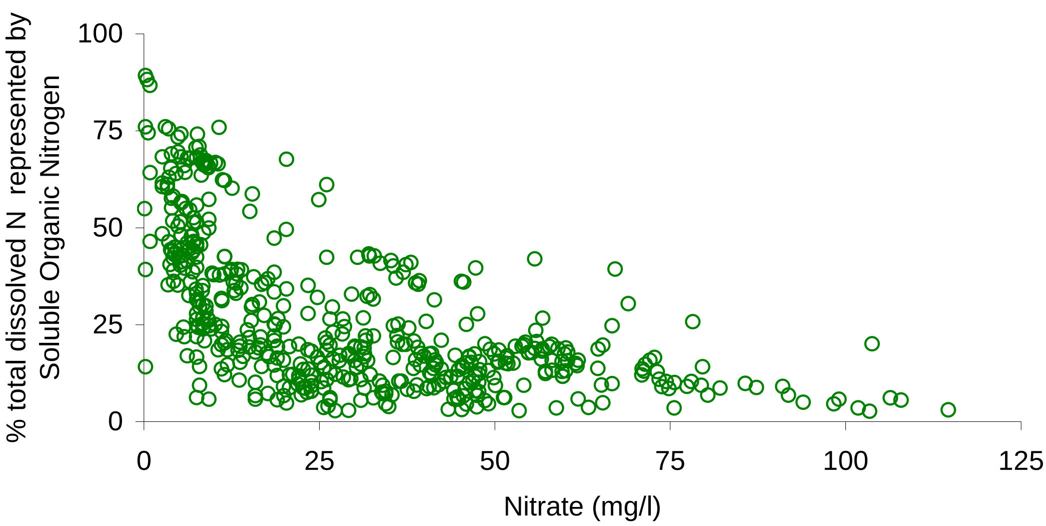


Figure 4.3 Relationship between nitrate-N concentration and the contribution of SON to total dissolved N. (Source: NIT18 project).

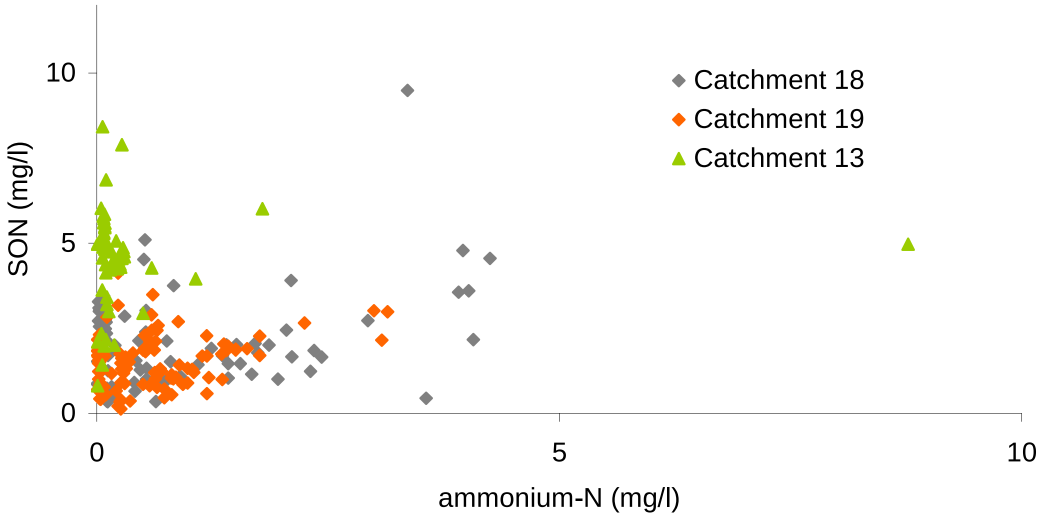


Figure 4.4 Relationship between ammonium-N and SON. (Source: NIT18 project).

The data presented in Figures 4.3 and 4.4 indicate that for the NIT18 micro-catchments, export of nitrogen as SON is relatively modest compared to that occurring as nitrate-N, and that animal wastes are a key source on some catchments. In catchments with high levels of nitrate, SON concentrations were generally less than 20% of nitrate-N concentrations. However, in catchments with smaller nitrate concentrations, some of which are designated NVZs for reasons of eutrophication risk, the SON concentrations can match or exceed the nitrate concentrations. It is well documented that SON can have an impact in sensitive estuaries (Badr *et al.*, 2008), or at least is an indicator of ecological status. Thus to neglect its contribution to the terrestrial and aquatic nitrogen cycles would seem unreasonable, though for compliance with the Nitrates Directive, it appears relatively unimportant. The question that needs to be considered further, but which is outside the scope of this project, is what is the fate of the SON lost from agricultural systems to watercourses - what does it contribute to nitrate concentrations, to nitrous oxide emissions, and to aquatic ecological change?

4.3 Faecal indicator organism (FIO) losses to water

The key risk factors for the loss of FIOs from agriculture are the slope of land and the proximity/connectivity to the watercourse (Kay *et al.,* 2010). Ongoing research in Wales is indicating that a great deal of progress has already been made in reducing uncontrolled flows from yards and hardstandings in priority catchments where funding has been made available for improvements. Similar positive evidence is emerging in England in conjunction with the CSF capital grant scheme which offers options for clean and dirty water separation and yard roofing. However, evidence from category 1 and 2 pollution incidents (pers comm., Environment Agency) suggests that this improvement is patchy with the total number of pollution incidents remaining steady over the passed five years.

Work by Stapleton *et al.* (2011) in Scotland found unexpectedly high microbial levels in SW derived from livestock farming activities. It was concluded that over half the microbial flux input was associated with high flow conditions (i.e. events over a threshold). The researchers concluded that connectivity with the watercourse at times of peak events is key to understanding the risk of microbial pollution, rather than the pressure as represented by the density of livestock farming *per se*.

4.4 Ammonia loss to air

Agricultural production systems are recognised as a major source of atmospheric ammonia (NH_3) (Sommer, 1995), and the spreading of slurry is one component of that source. Factors affecting ammonia emissions from manures are reviewed by Sommer and Hutchings (2001) and in Nicholson *et al.(*in press).

5 AGRICULTURE IN NVZS

5.1 Agricultural context of NVZs across England

The NVZs across England contain about 6.17 m ha, 69% of the national arable land, and about 2.5 m ha, 57% of the managed grassland. They contain about 60% of cattle, 34% of sheep, 83% of pigs, 81% of broiler hens and 69% of laying hens (Figure 5.1). The majority of nitrate loss within the NVZ area arises from arable land. (Source: Defra statistics mapped within the ADAS land use system).

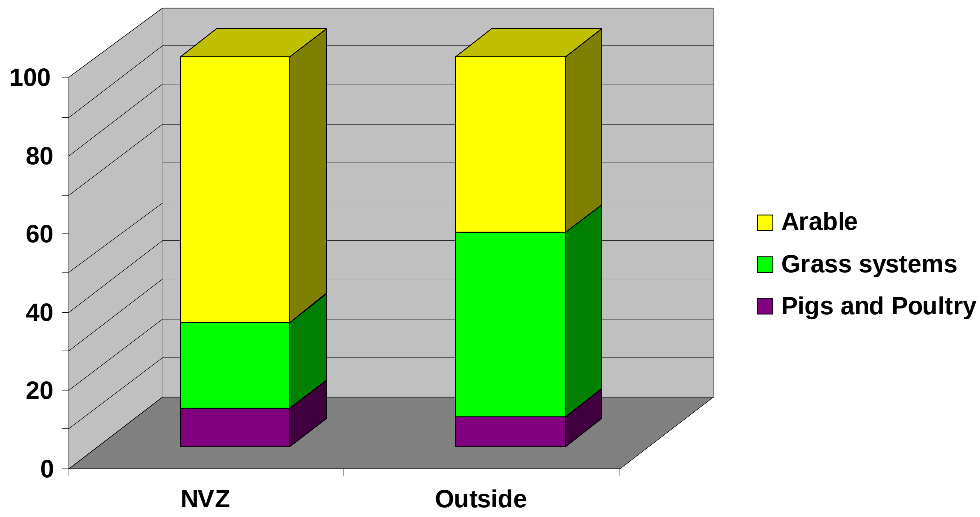


Figure 5.1. Proportion of nitrate loss arising from 3 main farming sectors, inside and outside NVZs. Estimation made by the ADAS NEAP-N model. (Source: NIT18 project).

The Nitrates Directive is implemented against a background of other underlying changes to agriculture, in addition to its own impacts on land management. Some of the relevant agricultural trends are discussed in this section.

The nitrogen surplus on grassland has been falling steadily for many years, largely due to reductions in manufactured fertiliser N inputs (Figure 5.2).

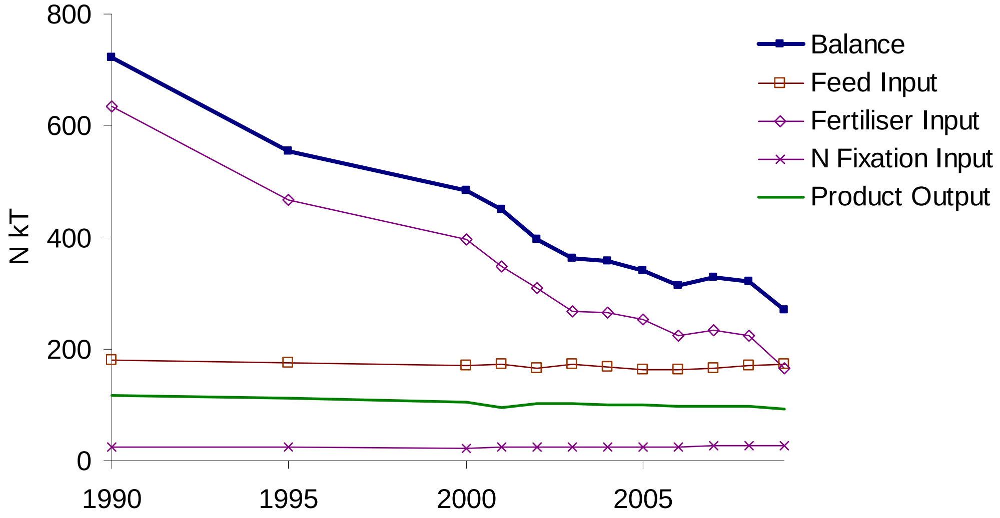


Figure 5.2. Trend in the components of the grassland sector N balance for England and Wales, showing the steady reduction in fertiliser N inputs. (Source: Johnson & Lord 2011, based on Defra statistics).

The reduction in manufactured fertiliser N inputs is only partly accounted for by changes in stock numbers and production, which have declined by only a small amount (Figure 5.3 and 5.4).

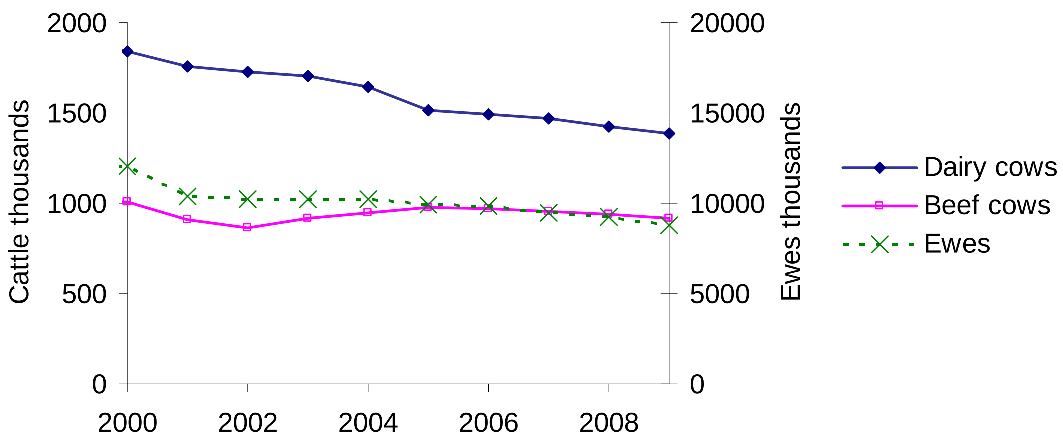


Figure 5.3. Trend in the numbers of grazing livestock in England and Wales, indicated by dairy cows, beef cows and ewes. (Source: Johnson & Lord 2011 based on Defra statistics).

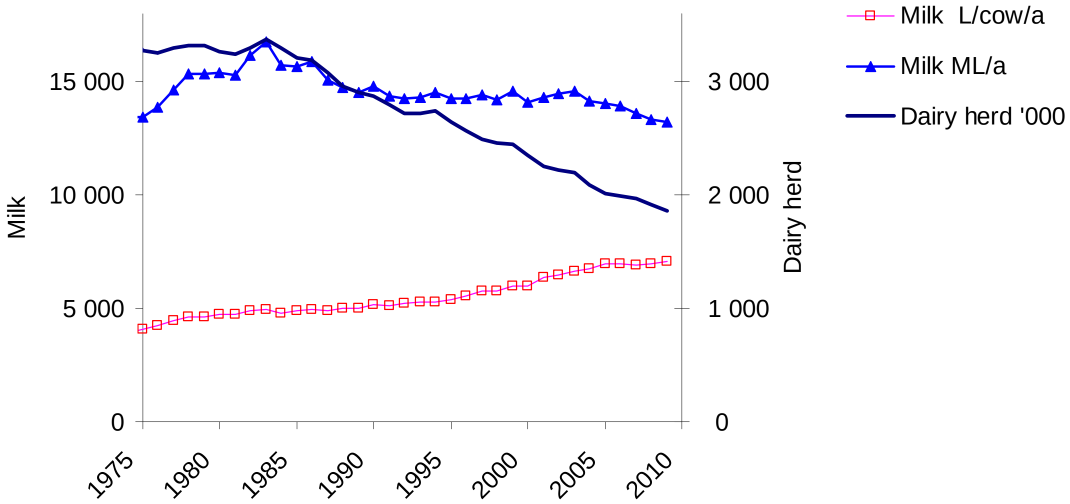


Figure 5.4. Trend in UK milk yield in relation to cow numbers and yield per cow (Source: Johnson & Lord 2011 based on Defra statistics).

Within the arable sector, manufactured fertiliser N inputs and the overall N balance have changed rather little in recent years (Figure 5.5). However, survey data have shown a steady improvement in the extent to which fertiliser N inputs are reduced where manures have been applied. (Source: BSFP). This can be attributed to the NVZ AP.

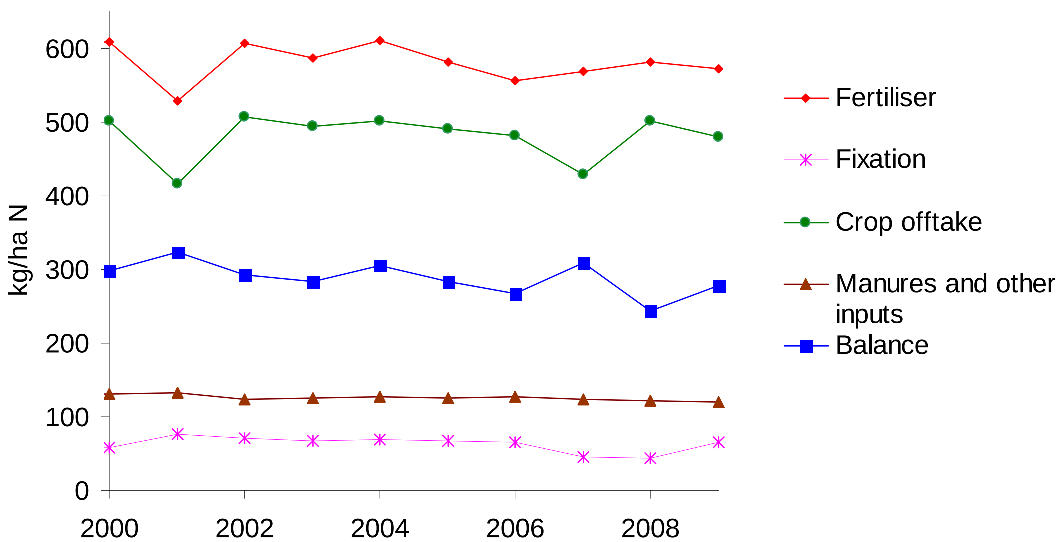


Figure 5.5. Harvested crops sector: Components of the net N balance (kg N per ha) for harvested crops. (Source: Johnson & Lord, 2011, based on Defra statistics).

In contrast, the P surplus in *both* grassland and arable sectors has fallen significantly over the past decade (Figure 5.6). The P surplus in the arable sector is now close to zero indicating that there is relatively little capacity for further reductions. There is still a significant surplus of P in the grassland sector, mainly due to the import of P in feed.

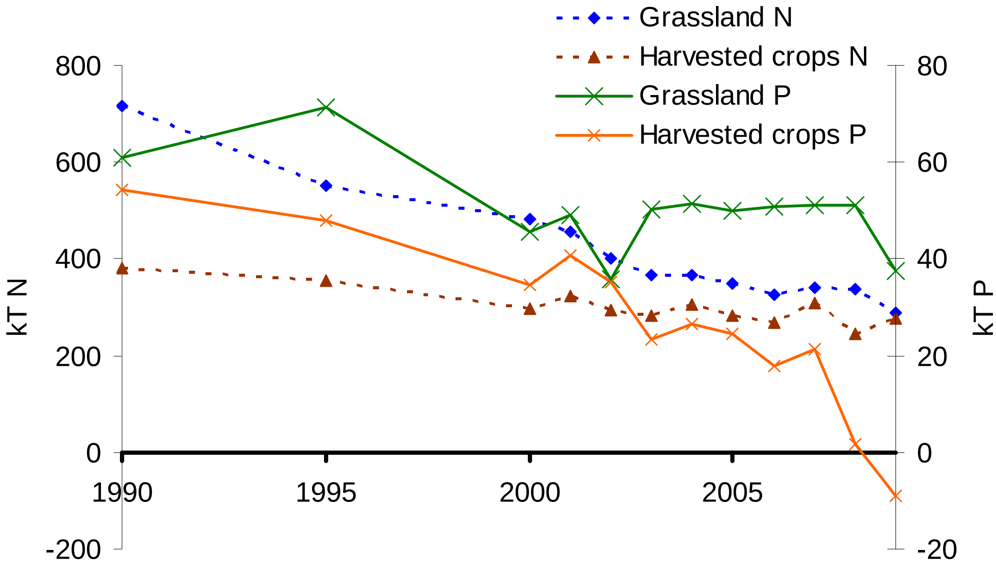


Figure 5.6. Trends in N and P surplus. (Source: Johnson & Lord, 2011 based on Defra statistics .)

Although N and P farmgate balances are useful for developing our understanding of trends in inputs and management practices within agricultural systems, they do not relate directly to nutrient losses to the environment. Agricultural systems that are in balance can still provide significant fluxes of nutrients to the environment. The national scale balances reported in Johnson and Lord (2011) can also obscure significant local variability. The farmgate balances need to be combined with evidence from national scale monitoring data and analysis (section 2), detailed monitoring data and analysis (section 3) and modelling (section 6), in order to develop and maintain a compelling 'weight of evidence' on the current status of, and trends in, nutrient losses from the agricultural sector across England and Wales.

5.2 Changes in agricultural practice within NVZs

Agricultural land use and management surveys, including the annual June Survey of Agriculture, the British Survey of Fertiliser Practice (BSFP), and the Farm Practice Survey (FPS), provide information on land management changes, some of which are relevant to nitrate loss and corresponding water quality.

It should be noted in assessing these data, that the NVZs occupy mainly the drier southern and eastern areas of England, and therefore contain a high proportion of national area of arable land. The proportion of the national area of extensive grassland in NVZs is low relative to the area outside NVZs.

Adjustment for N supplied by manures
The NVZ AP requires fertiliser N inputs to be adjusted to allow for other sources of N including manures. The average allowance made for N supplied by manure increased substantially following implementation of the NVZ AP. Taking data for winter wheat (the arable crop with the most robust data), the average adjustment increased from 8 kg/ha in 2003 to 30 kg/ha in 2006, but has since declined somewhat. The increase was greater, and was evident earlier, on land inside compared to outside NVZs (Defra, 2008). Similar trends are evident for other crops. National data suggest that there may now be some reversal of this initial improvement in adjustments, possibly related to the change in emphasis in the

regulations from use of recommendation systems, to compliance with farm-level calculations. The overall change in fertiliser N inputs to arable crops is small. Fewer than 20% of arable crops receive manure in any one year. A substantial reduction in N inputs to arable crops in 2008 was related to steep price fluctuations, but 2010 inputs averaged close to those in 2000-2002.

On farms within the NIT18 project, reductions in fertiliser N input where manure was used were observed for all major crop types. On average, fertiliser N inputs to winter wheat receiving manure in the same year were reduced by 28 kg/ha N. Similar reductions applied to other crops. This is close to the adjustment that would be recommended, taking into account factors such as the range of manure types and timings and soil type.

On grassland, this effect is difficult to assess mainly because the farms which apply most manure are likely to be those with the greatest stocking density and where fertiliser N inputs are therefore also high in order to provide sufficient grass growth. While fertiliser N inputs to grassland have been falling for many years, and continue to do so, it is difficult to assess any additional impact of the NVZ AP on the basis of this trend.

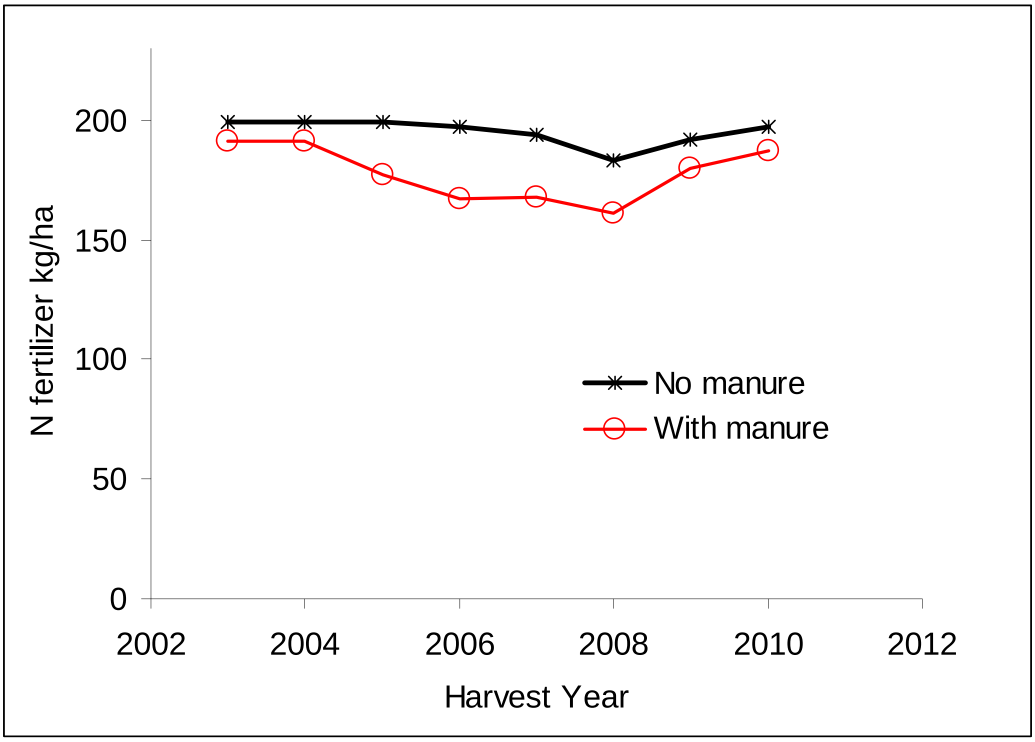


Figure 5.7. Mean application rates (kg/ha N) of manufactured fertiliser N on winter wheat, with and without manures in the same crop year. Britain. Source: BSFP

Overall, however, average fertiliser inputs to arable (tillage) land have changed little in the past decade, while inputs to grassland have fallen (Figure 5.8). The dips in

2008-9 were related to steep increases in fertiliser prices. These overall trends appear to be independent of the NVZ AP.

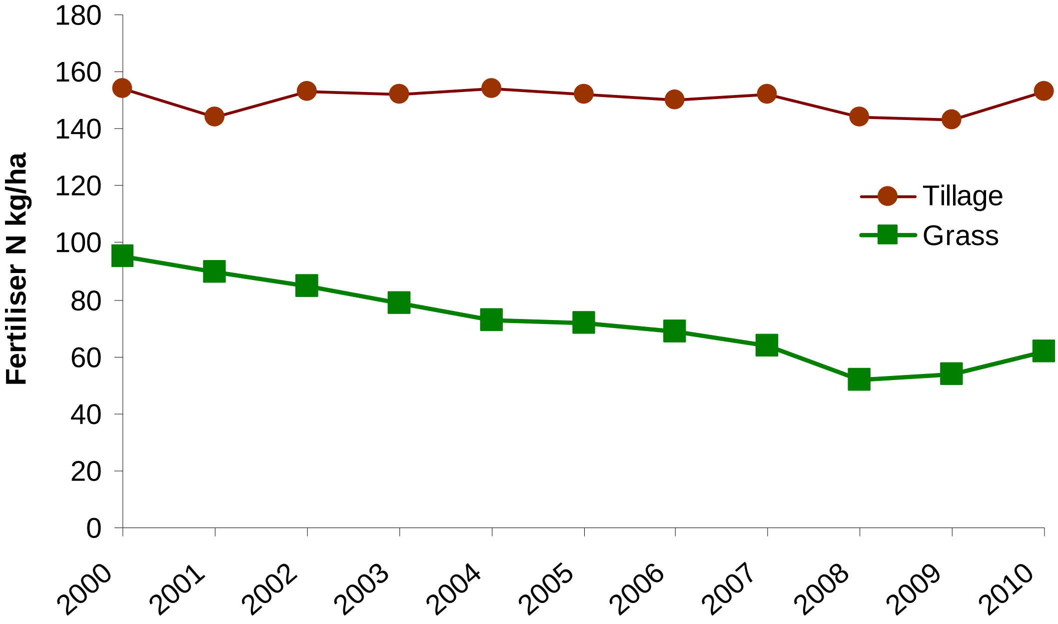


Figure 5.8 Mean application rates (kg/ha N) to tillage and grassland, England and Wales. Source: BSFP

Timing of manufactured fertiliser N applications

The BSFP indicates that about 1% of all fertiliser N is applied in the autumn. While inputs were greater in the 1980s, this proportion has been low and fairly stable recently. Very little fertiliser N is now applied in this period except to crops with a specific economic response, such as aftermath grazing on grass, and winter oilseed rape. This is reflected in the fact that most autumn fertiliser N is applied in August, when such crops can make most effective use of it.

Nutrient management plans

The proportion of farmers with nutrient management plans is increasing nationally, and this increase is particularly evident within NVZs (Figure 5.9). A greater proportion of farms within the original 1996 NVZs have Nutrient Management Plans compared to those within NVZs designated in 2002. Further clarification was introduced into the 2007 Farm Practices Survey, suggesting that, of those with no formal plan, around 60% did not grow crops (many being extensive beef or sheep farmers) and thus had less need for a nutrient management plan.

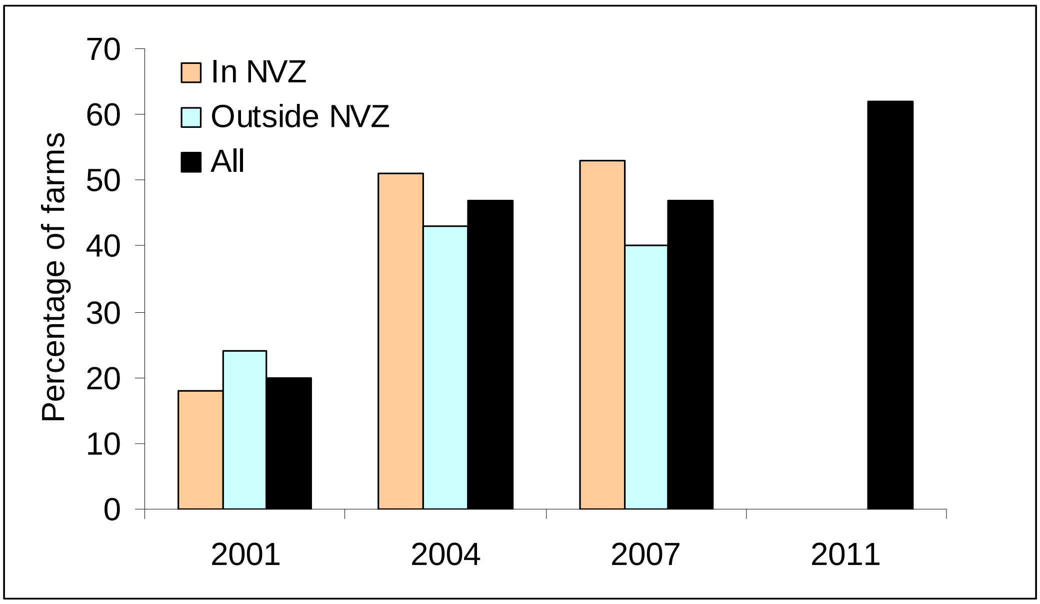


Figure 5.9 Proportion of farms with a Nutrient Management Plan (Source: Farm Practice Survey spatially re-analysed by Defra. M. Thomas, pers. comm.)[1]

Slurry storage

Average slurry storage has increased slowly over the past 10 years. The proportion of farms with less than 1 month's storage has fallen, but there remain a considerable number of holdings with insufficient storage to cover the NVZ Closed Period (Defra 2007, 2010) and (Figure 5.10).

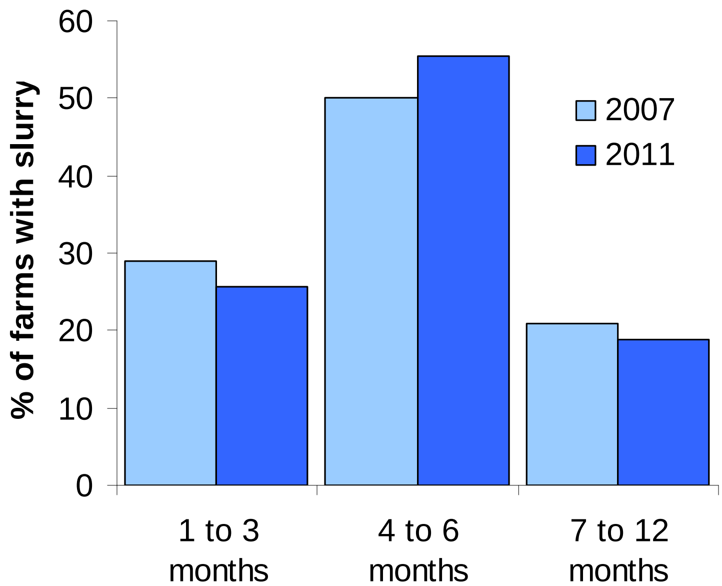


Figure 5.10. Slurry storage capacity (months). Source; Defra 2007, 2011.

[1] In 2001 recorded as farms with farm waste management plans. In 2004 recorded as holdings with manure management plans.

The 2007 survey found that storage was greater on arable farms, and within NVZs. Since NVZs have a greater proportion of arable land than the remainder of England and Wales, it is not clear whether this additional storage is due to the pressures of the NVZ AP or to the more limited times at which applications are practicable on arable crops. About 20 per cent of respondents to the Farm Practice Survey 2006 had made changes to their slurry storage during the previous 2 years, and change was slightly more likely on farms inside NVZs than outside. The most common change was to divert clean water away from an existing store. Farms within NVZs were more likely to have increased the size of their slurry store or to have provided additional storage in a new structure.

6 MODELLING OF WATER POLLUTION FROM AGRICULTURE: NITRATE AND PHOSPHORUS

6.1 Modelling approach

A framework consisting of modularised process-based field scale models was developed as one component of the NIT18 project, in order to assess the impacts of agriculture on water quality. The framework was designed to be:

- sensitive to relevant management practice, weather and soils
- responsive to simple inputs available from basic survey data
- representative of multiple pollutants
- operational on a daily time step
- scalable to catchment or national level simulations

Field scale model
The basic structure of the field scale version of the ADAS Pollutant Transport (APT) framework is depicted in Figure 6.1. The framework utilises basic input data on field management and local weather to drive both the phosphorus and sediment (PSYCHIC) and nitrate (NIPPER) models, as well a shared water balance model, such that the framework is driven by a common hydrology.

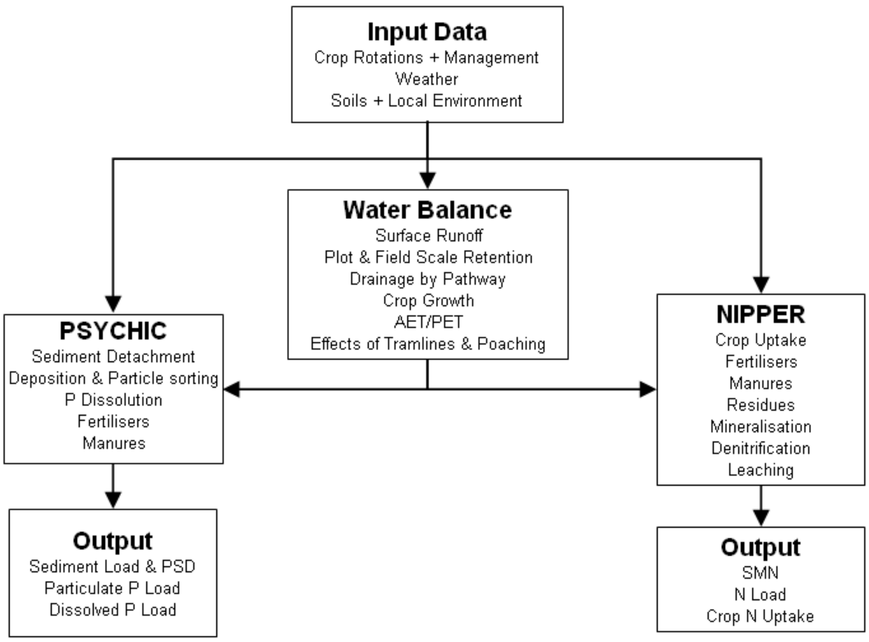


Figure 6.1. Schematic for the new integrated APT modelling framework (field scale).

The major concept within the new framework is that the volume of flow generated, and the pathway that this flow takes, is the main determinant of water quality. A schematic of how flows are conceptualised in the APT framework and the scales of operation considered is shown in Figure 6.2.

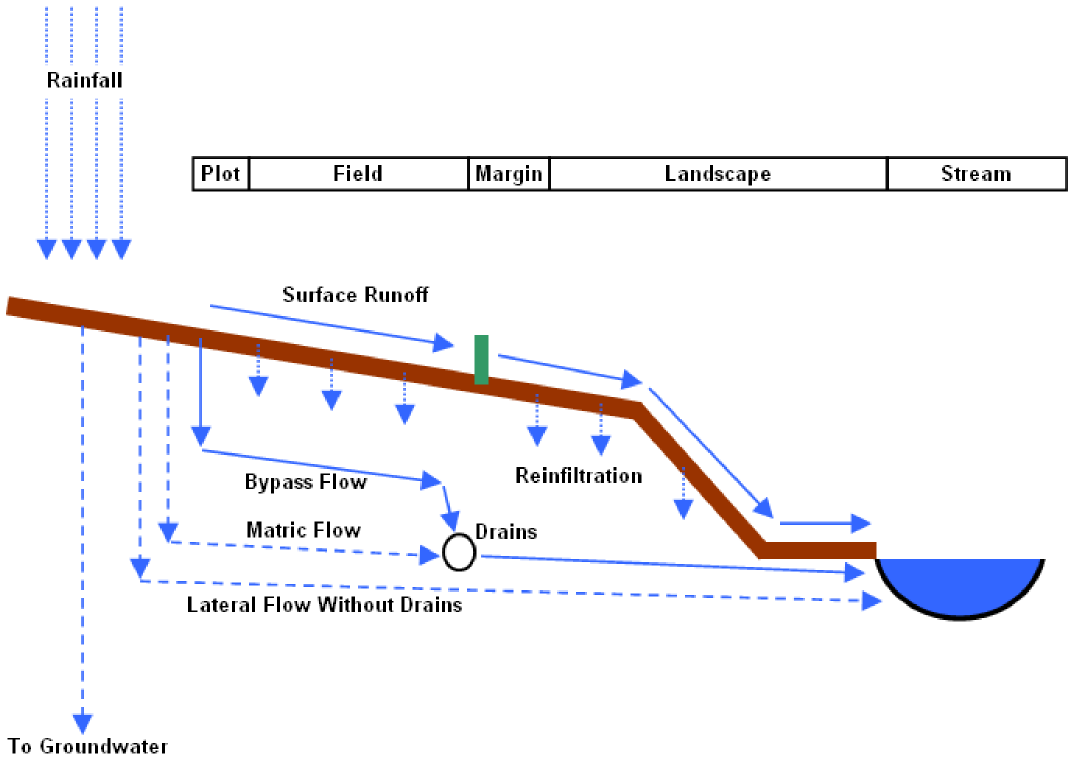


Figure 6.2: Flow pathways and delivery scales within the APT model.

Figure 6.2 shows how flow is routed between plot and watercourse in the framework. Flow is generated at plot scale (assuming a notional plot length of 10 m) and pollutants are mobilised through the different pathways. Reinfiltration and deposition of pollutants can occur within field (up to the margin feature), at the margin feature (outside field) and then between field and stream. Any change in flow and pollutant concentration that occurs between field and stream is considered in a separate landscape module used in the catchment scale version of the APT model.

Flow and pollutant mobilisation, transport and delivery within the APT framework are dependent on a number of field and landscape scale factors, including:

- Soil type and state of ground
- Crop cover and protection
- Cultivations
- Presence or absence of tramlines
- Livestock intensity
- Manure type, timing and method of application

Full model documentation for the hydrology and sediment components of APT will be delivered under Defra project WQ0128.

Catchment scale model

The basic structure of the catchment scale version of the framework is depicted in Figure 6.3. The framework relies on a range of census and survey data to create suites of field scale input files, based around crop rotations, which are assumed to represent the agriculture occurring within each catchment. The field scale outputs from all the different field scale simulations for a catchment are then subject to landscape retention, and the results area-weighted to reflect the different land cover in each 1km^2 within the catchment.
The data used to create the catchment scale inputs include:

- WFD sub-catchment boundaries

The model framework has been designed to operate at any spatial scale, but the inputs have been generated for WFD sub-catchments. Uncertainty in the underlying land use and livestock data suggests a minimum resolution of ca. 25 km^2, so sub-catchments have been amalgamated where they are below this threshold, resulting in 3,400 individual sub-catchments for a complete national simulation for England and Wales.

- Soils

Within each WFD sub-catchment, the soil series (based on Natmap 1000 1 km^2 detailed soils data for England and Wales (NSRI)) are amalgamated into groupings based on the HOST classification (Boorman *et al*., 1995), with each group represented by the most important soil(s) within that group in the specific sub-catchment.

- Land use data

The field scale model operates with 18 crop categories (including woodland and rough grazing), whereas the annual agricultural census has a variable number of categories (up to 46). The census data are cross-mapped against the modelled crop categories in order to produce layers suitable for input into the model.

- Land cover

Estimates of land cover (arable, grassland, rough grazing and woodland) are more robust at 1 km^2 resolution than the land use data. These data are thus used to disaggregate the model outputs for each WFD sub-catchment, utilising the proportions of each soil group / land cover combination within each 1 km^2. This allows for more detailed data to be used for the calculation of landscape retention.

- Rotation calculator

Cropping data for each soil group in each catchment is converted to a series of three-year rotations.

- Manure and excreta data

Livestock and cropping data by WFD sub-catchment polygon are used as input for the Manures GIS system (Defra project WQ0103) to produce a relevant layer of manure and excreta data. This is assumed to be distributed evenly across soils within the sub-catchment.

- Manure allocator

The manure assigned to a soil is allocated to the most appropriate crops in the rotations and grassland areas, based on a series of expert rules (e.g. preference for cut grass then grazed grass; preference for 2nd cereals following break crops over 1st cereals).

- Lookup tables

The model inputs rely on a number of additional inputs, including crop husbandry parameters (based on expert knowledge), and fertiliser application rates (based on RB209) and timings (based on BSFP analysis).

- Field statistics, boundary features

Information on the average size of a stratified sample of fields was available from Defra project PS2233. This stratification was by Aquatic Landscape, so a 1 km^2 version of the Aquatic Landscapes dataset was used to distribute the statistics across England and Wales. Data from the Countryside Survey (CEH, 1998) were used to obtain information about boundary types across England and Wales, with a 1 km^2 dataset of landscape types available from the Countryside Information System used to distribute the survey results across England and Wales.

- Slope and altitude

Average slopes and altitudes are estimated for each 1 km^2 cell from the CEH hydrologically-corrected DEM at 50 m resolution.

- Atmospheric deposition

A dataset of atmospheric nitrogen deposition provided by CEH for a recent and typical emissions year (2003) is used.

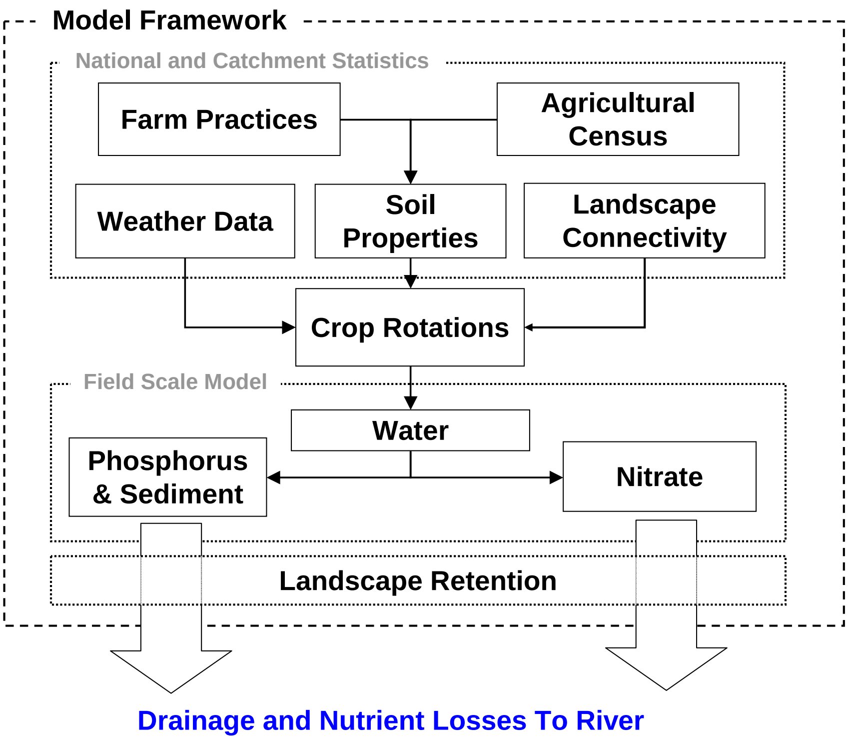


Figure 6.3: Schematic for the new integrated APT modelling framework (catchment scale).

6.2 Spatial variation in pollutant loss and impact of measures

The NVZ AP measures have variable impact depending on both the effectiveness of the measure itself, and the extent to which it is applied. The majority of the NVZ measures will be most effective in areas with high livestock numbers. For example, the Closed Period legislation will most affect dairy farms and pig farms which produce slurry; poultry farms; and surrounding areas which apply these manures. The N loading of these manure types is greatest in dairying areas of the west, and in parts of East Anglia and Humberside (Figure 6.4).

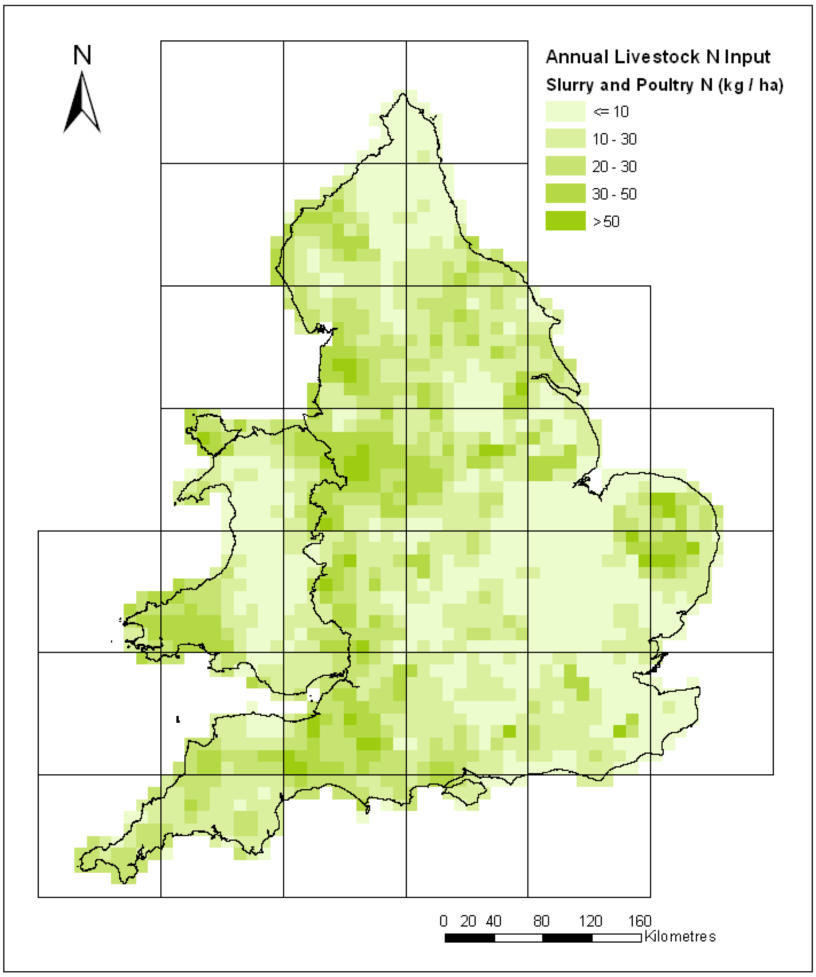


Figure 6.4. N loading of slurries and poultry manures (kg N per ha of agricultural land).

The requirement to comply with fertiliser recommendations will have most impact on arable farms which use manures, since these were the areas where compliance was previously poorest. Figure 6.5 shows total N loading as manures of all kinds, giving an indication of the areas where this measure will impact most. The farm-scale limit on excretal returns affects mainly dairy farms, which are concentrated in the west of the country.

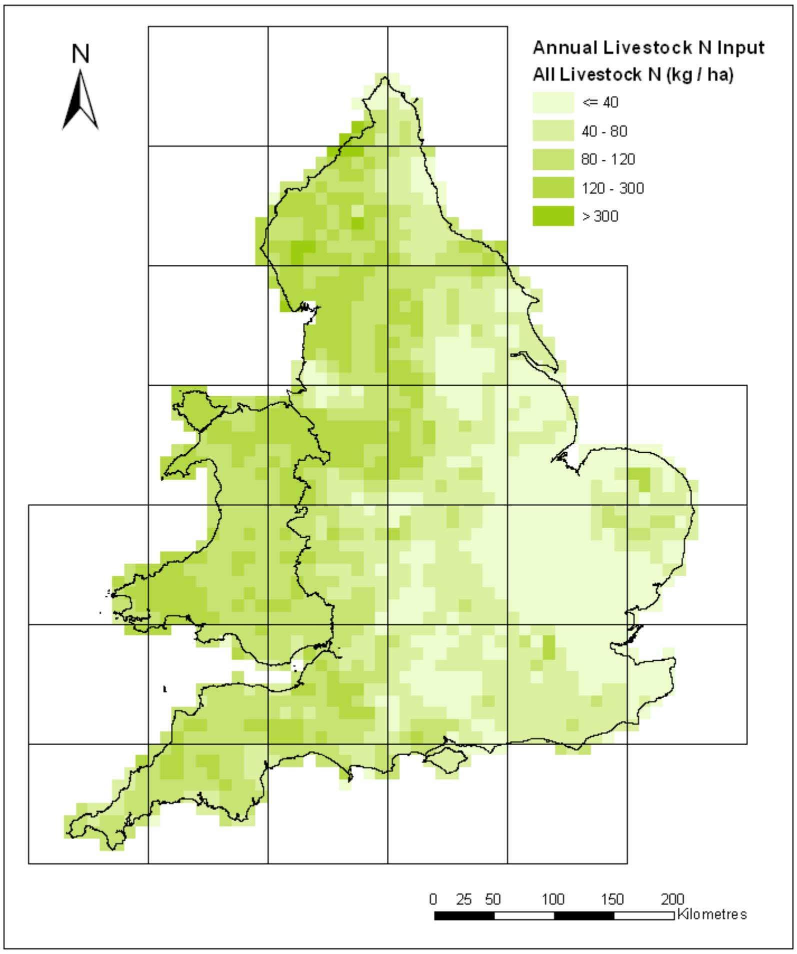


Figure 6.5. Total N as livestock excreta (kg N per ha agricultural land).

As a result of these distributions, the impact of the current NVZ AP will have been highly variable, and there are significant areas where little impact is to be expected.

6.2.1 Spatial pattern of the impacts of cover crops

The catchment-scale model uses spatial data on cropping, livestock numbers, soils and climate to generate spatially-distributed estimates of pollutant loss and concentration. To assess the spatial impact of cover crops on nitrate, phosphate and sediment, the model has been run with national datasets for current land use, and then for current land incorporating cover crops where appropriate.

Impacts are greatest where the proportion of spring sown crops suitable for cover cropping is greatest, that is, on light soils in areas of arable cropping.

Impact of cover crops on nitrate losses.
The average reduction of nitrate loss from agricultural land to water, over the whole NVZ area, is 7% with a range of 2 – 9%. However, if we consider the Groundwater NVZ area, where permeable soils and spring cropping are more prevalent the average reduction in GW increases to 8% with a range of 4 – 10%.

The impact of cover crops on nitrate loss is greatest in areas of light soils such as Norfolk and Shropshire, where spring cropping is common, Figure 6.6. There is little impact in areas of heavy clays whether they are under grass or arable land use, due to the small area of spring cropping suitable for cover crops.

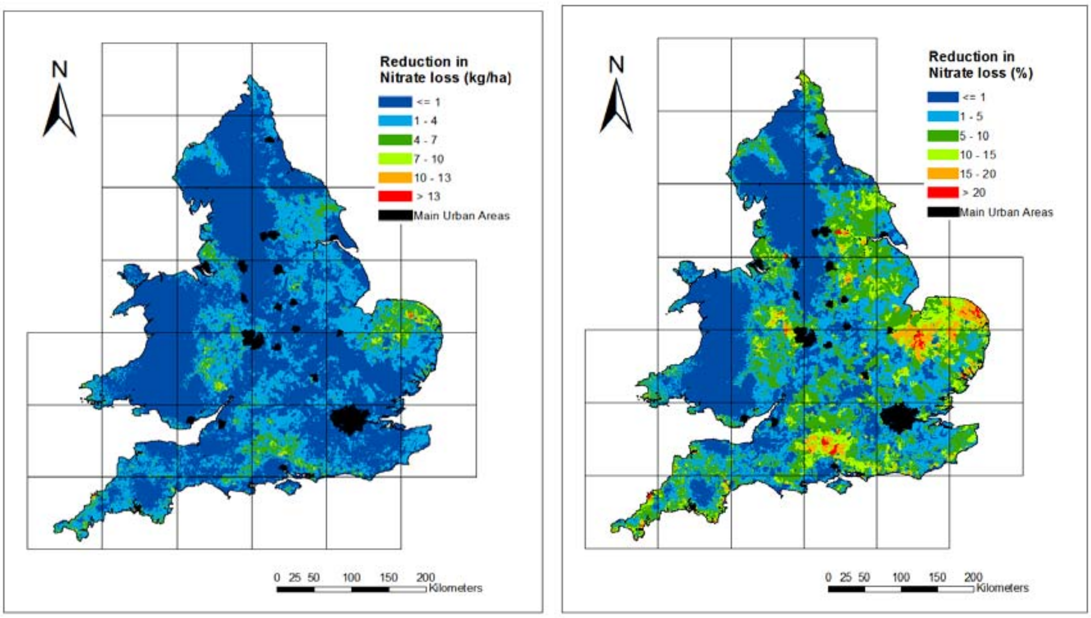


Figure 6.6. Modelled spatial distribution of change in N loss as a result of introducing compulsory over-winter cover crops where land would otherwise be bare.

Impact of cover crops on phosphate losses.
The average reduction of P over NVZ area is 0.3 % with a range of 0 – 1%. However, if we consider the GW NVZ area, where permeable soils and spring cropping are more prevalent, the average reduction increases to 0.5% with a range of 0 – 3%.

The impact on P is due to reduction in surface runoff risk. This is particularly great in western areas due to the greater rainfall (Figure 6.7). The greatest impact was during the autumn and early winter period. It is important to use the maps of percentage change and absolute change together in order to fully understand the spatial pattern of the impacts. The greatest percentage change is in East Anglia, however, P losses there are low.

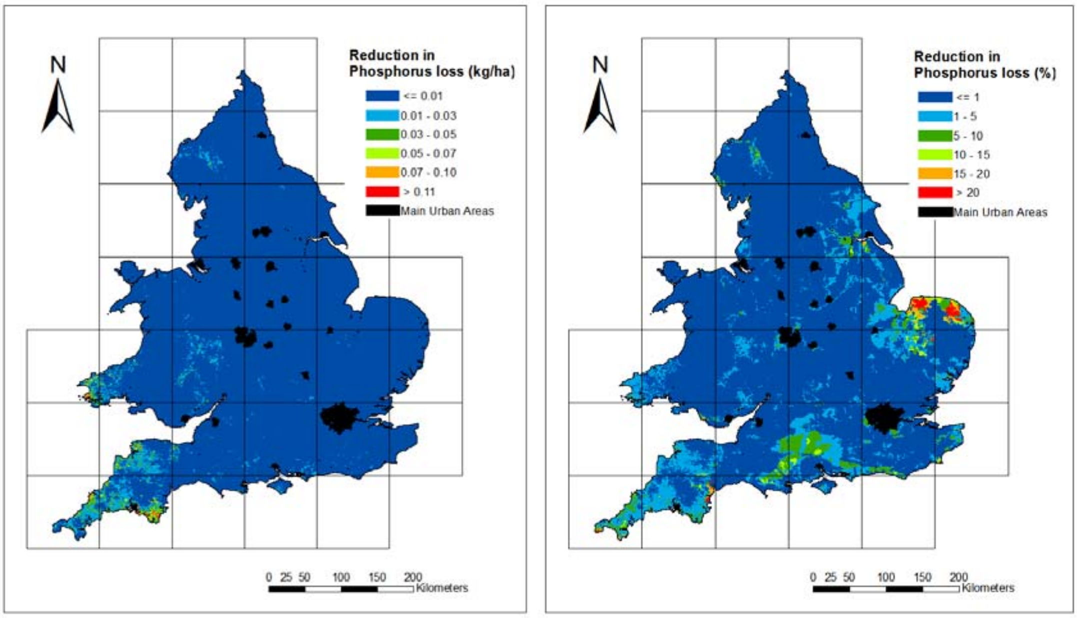


Figure 6.7. Modelled spatial distribution of change in P loss as a result of introducing compulsory over-winter cover crops where land would otherwise be bare.

Impact of cover crops on sediment losses.
The average reduction of sediment over NVZ area is 1% with a range of 1 – 2%.

Mapped sediment loss shows a very similar pattern to mapped P loss, as is to be expected since largely the same causal factors are operating. Impacts will be greatest in areas of high runoff risk and high rainfall, and where spring cropping occurs (Figure 6.8). Again it is important to use the maps of percentage change and absolute change together in order to fully understand the spatial pattern of the impacts.

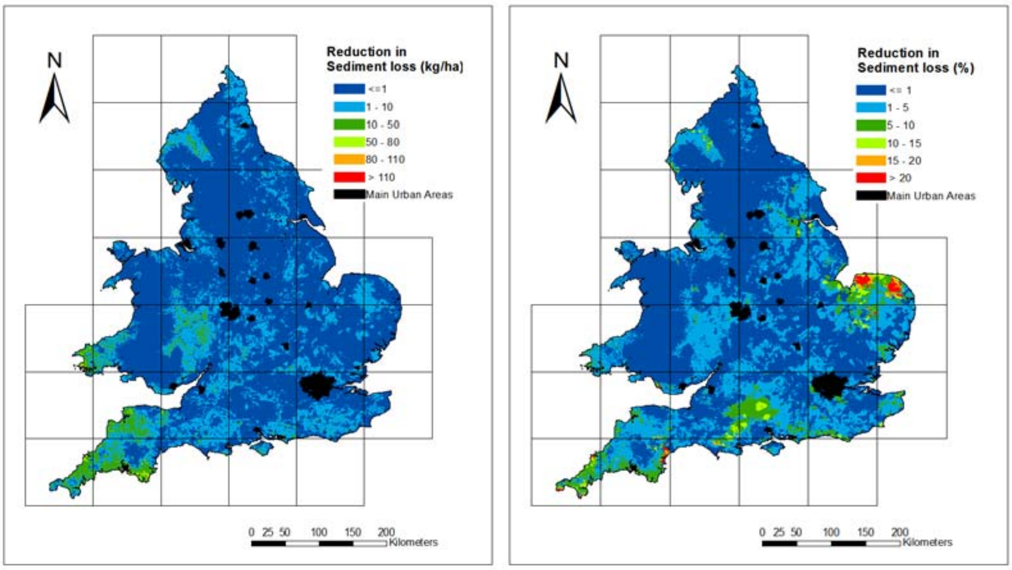


Figure 6.8. Modelled spatial distribution of change in sediment loss as a result of introducing compulsory over-winter cover crops where land would otherwise be bare.

6.2.2 Commentary on spatial modelling

The use of a daily multi pollutant model (APT) for national scale modelling is ambitious and presents a number of technical and practical challenges. Model run times are typically three days per scenario with data output files running to many Gb per model run. Note that while model predictions give a good picture of the national impact, the values at any given point are subject to considerable uncertainty due to:

- Data uncertainty (spatial imprecision). The data used to assess the location of crops, and management are derived from surveys, so represent typical rather than locally precise information. Weather also varies from year to year. However at catchment scale and over a number of years, these variations will tend to average out.
- Model or structural uncertainty. We may have incorrect representation of the physical processes at any given location perhaps because a particular catchment is atypical.
- Parametric uncertainty. The value of parameters used in the model to represent soils, crops etc are based on experimental data which always include uncertainty. Monitoring and experimental results used to validate the model are also variable and the science is constantly developing. This is a particular issue with sediment data.

However at catchment scale, the local variability tends to average out, and the data give a reliable indication of potential impacts.

6.3 Relationship between field scale impacts and those measured at catchment scale

The designation of NVZs, and assessment of progress towards improved water quality, are based on monitoring of ground and surface waters at the EA water quality monitoring points. Key considerations include:

Land use and the NVZ AP measures

- A given reduction in nitrate loss from agriculture may have a smaller percentage impact at these monitoring points if there is a substantial contribution from other non agricultural sources, (e.g. urban, forest).

- The impact of a measure at catchment scale is a combination of its effectiveness per hectare when used, and the proportion of the total area of the catchment where it is applied. For example, compliance with fertiliser N recommendations has relatively small effects on most farms, but has a significant overall impact because it is widely applicable. The Livestock manure N farm limit of 170 kg N/ha can have a substantial impact but affects only a small proportion of the land area.

- Impacts of the AP measures are greatest in areas with livestock, particularly arable areas with pigs or poultry, and intensive dairying areas. This is because most of the measures either directly address livestock systems (e.g. closed period for application of manures) or have greatest impact where manures are regularly used (e.g. compliance with N recommendations). In some areas, measures will have a significant impact while in others, the impact will be negligible.

Issues specific to groundwater catchments

- In groundwaters, and groundwater-fed catchments, responses to changes in farm practices may be delayed by many years because of the time taken for water to move through the unsaturated zone to the aquifer (see Figure 6.8). The delay in leachate reaching groundwater abstraction points is highly variable, ranging from weeks to decades or more. A broad classification can be made to indicate areas where greatest delay is expected – these include many of the major potable water sources (Williams *et al*., 2011).

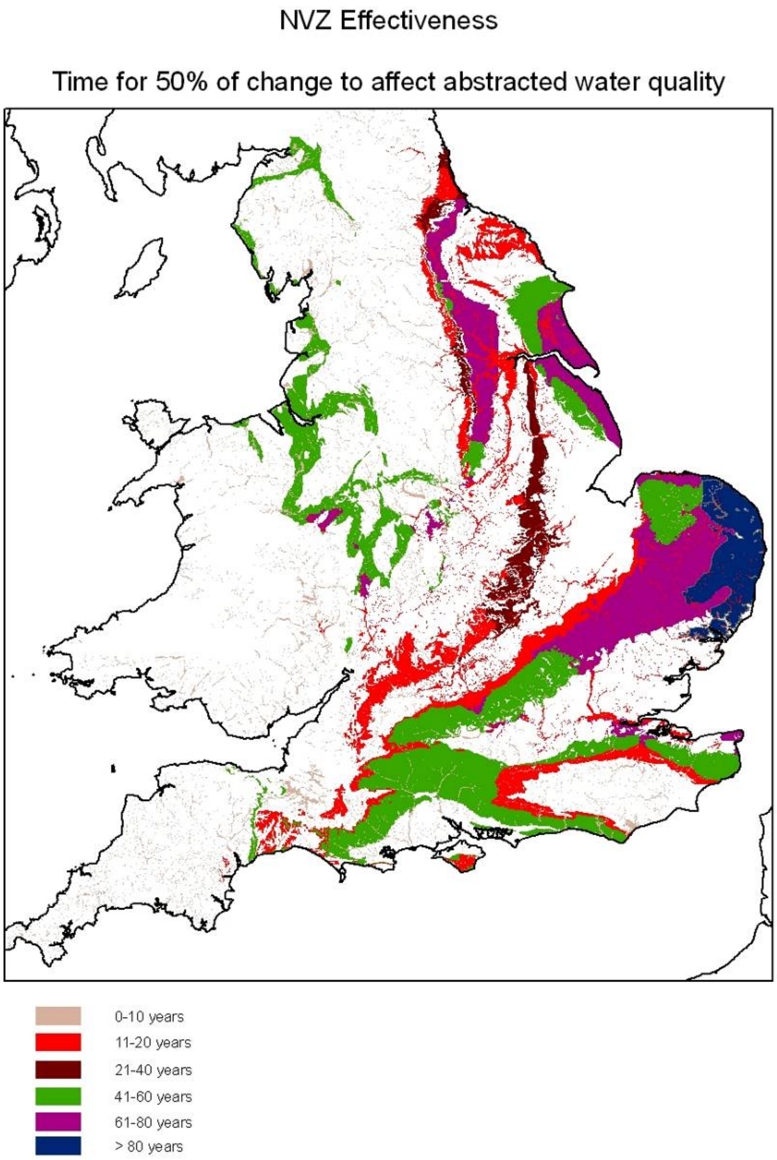


Figure 6.8. Typical delay times between the change in nitrate leaching from agricultural land, and the measured change at the groundwater abstraction point. Source: British Geological Survey for the NIT18 project. Williams *et al*., 2011.

Issues specific to surface waters

- Nitrate concentrations in surface waters respond rapidly to changes in farm practices. For this reason, nitrate concentrations are very variable day to day. Within surface water catchments, designation is based on the upper 95th percentile nitrate concentration (Q), which is usually greater than the annual mean nitrate concentration (M) by a factor between 1.4 and 2. Where this ratio approaches 2, a 95th percentile concentration below 50 mg/l nitrate requires an average concentration of below 25 mg/l nitrate, which is lower than achieved in most arable cropping areas.

The ratio of the 95th percentile to the mean is greatest in 'quick response' catchments (i.e. those that respond rapidly to rainfall events) such as those dominated by heavy clay soils that predominantly drain by 'bypass/crack' flow and where effective field drains are present. These watercourses are described as having a low Base Flow Index (BFI) in contrast to groundwater-fed streams, which have a high BFI (Figure 6.9). The ratio is also lower in urban catchments, because of sewage inputs which reduce the differential between winter and summer nitrate concentrations.

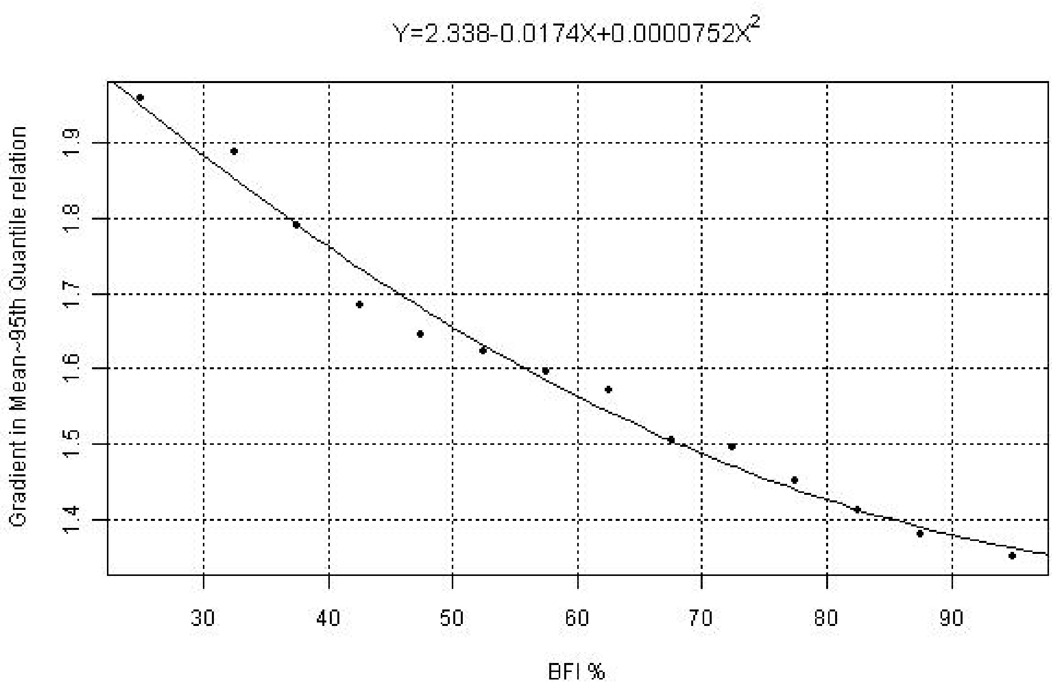


Figure 6.9. The ratio of the upper 95th percentile nitrate concentration to the mean nitrate concentration as a function of catchment Base Flow Index (BFI). Source: EA data analysed within the NIT18 project.

The ratio of the 95th percentile (Q) to the mean (M) nitrate concentration can be predicted by:

$Q/M = 2.338-0.0174BFI+ 0.0000752(BFI)^2$
where BFI = Base Flow Index

There is also a predictive relationship with the percentage of catchment under urban land use:

$Q/M = 1.86 -0.024 * U + 0.00041 * U^2$

where U = urban%

This is most likely due to contributions from sewage treatment works, which tend to increase nitrate concentrations in summer, thereby countering the usual tendency for concentrations to be greater in winter.

Given the wide day-to-day fluctuations in nitrate concentrations in watercourses, different ways of averaging the data can have different results. Averaging of all EA measurements gives a time-weighted mean, whereas modelled results generally give a load and a quantity of water, from which a flow-weighted mean concentration can be estimated. Nitrate measurements made at high flow will make a greater proportional contribution to the flow-weighted mean. It might be expected that these two values would differ, and that since concentrations in agricultural catchments tend to be greater in winter when flows are high, the flow-weighted mean concentration would be the greater value. This has indeed been demonstrated for small catchments. However, it was found that for moderate to large catchments (those within the Harmonised Monitoring Stations scheme) the two values are very closely related, and on a 1:1 line (figure 6.10). Thus, modelled mean nitrate concentrations are directly comparable with site means published by the EA. From the mean, the 95th percentile can then be estimated using the relationships above.

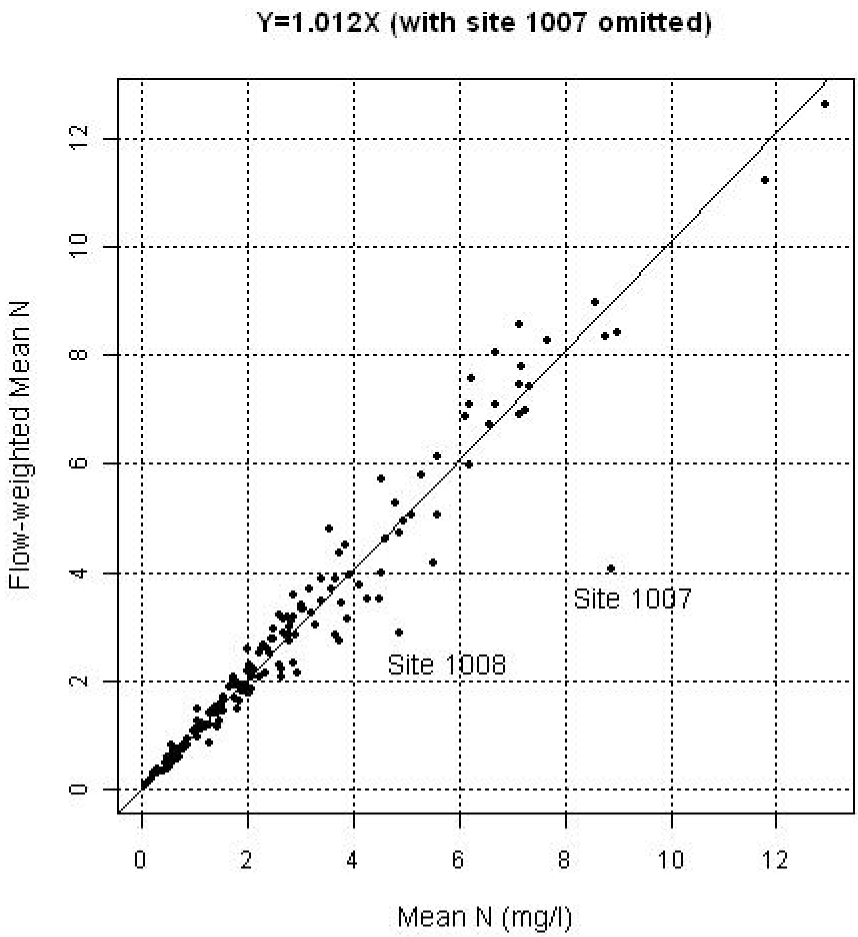


Figure 6.10. Relationship between simple sample means (time-weighted means, x axis), and flow-weighted mean nitrate concentrations at Harmonised Monitoring Point stations in England and Wales. Source: EA data analysed within the NIT18 project.

7 CONTRASTING THE ENGLISH NVZ AP WITH THOSE OF SIMILAR COUNTRIES IN NORTHERN EUROPE.

7.1 English obligations under the Nitrates Directive.

The main NVZ AP measures in England that came into force on 1st January 2009 are summarised below:

Storage of organic manure (including slurry)

If a farm stores any organic manure (other than slurry), or any bedding contaminated with any organic manure, it must be stored in a vessel, covered building, on an impermeable surface or on a temporary field site, as long as the material is solid manure that can be stacked in a free standing heap and that does not drain liquid.

If the farm produces slurry it must provide sufficient storage for all slurry produced on the holding during the storage periods. A slurry store must have the capacity to store, in addition to the manure, any rainfall, washings or other liquid that enters the vessel (either directly or indirectly) during the storage period.

Limiting the application of organic manure

In each calendar year, the total amount of nitrogen in livestock manure applied to agricultural land, whether directly by the animals whilst grazing or by spreading, should not exceed 170 kg multiplied by the area of the holding in hectares. Within any twelve month period, the total amount of nitrogen in organic manure spread on any given hectare on the holding must not exceed 250 kg.

Planning the spreading of nitrogen fertiliser (includes manufactured fertiliser, slurry and other organic manures)

The NVZ rules for this aspect relate only to planning and record keeping.

Total nitrogen spread on a holding

The farmer must ensure that the total amount of:

a) nitrogen from manufactured fertiliser, and
b) nitrogen available for crop uptake from livestock manure

spread on the crops, does not exceed the limits set out in the guidance for any 12 month period. For the purposes of this measure, the farmer must first establish the total amount of nitrogen in the manure, either using standard figures or by sampling and analysis.

Controlling the spreading of nitrogen fertiliser (includes manufactured fertiliser, slurry and other organic manures)

The farmer must not spread nitrogen fertiliser on land if there is a significant risk of nitrogen getting into surface water, in particular taking into account; the slope of the land, any ground cover, proximity to surface water, weather conditions, soil type and presence of land drains. The farmer must not spread nitrogen fertiliser if the soil is waterlogged, flooded or snow covered, or has been frozen for more than 12 hours in the previous 24 hours. No manufactured fertiliser should be applied within 2 m (for organic manures 10 m) of surface water.

Organic manure must not be spread within 50 m of a borehole, spring or well and nitrogen fertiliser must be spread in as accurate a manner as possible. If the farmer applies organic manure on to the surface of bare soil or stubble (other than soil that has been sown) they must ensure that it is incorporated into the soil in accordance with the guidance.

All slurry must be spread using equipment with a low spreading trajectory, i.e. below 4 m from the ground.

Closed periods for spreading nitrogen fertiliser (includes manufactured fertiliser, slurry and other organic manures)

Organic manure with high readily available nitrogen must not be spread on land during the following inclusive dates (the "closed period"):

Soil type	Grassland	Tillage land
Sandy or shallow soil	1 September to 31 December	1 August to 31 December
All other soils	15 October to 15 January	1 October to 15 January

Manufactured fertiliser must not be spread on land during the following periods (all dates inclusive):

a) on grassland, from 15 September to 15 January
b) on tillage land, from 1 September to 15 January.

Organic producers may spread organic manure provided that each hectare on which organic manure is spread does not receive more than 150 kg total nitrogen between the start of the closed period and the end of February. There are a small number of crops which can receive fertiliser during the closed period and these are listed in the guidance.

7.2 The current position in Europe

> *For decades, Western Europe has had the highest total fertilizer N use and animal N excretion per unit agricultural land in the world. Both started to level off from the second half of the 1980s. A series of environmental policies have been launched in the EU from the early 1990s onwards to decrease these losses*. (Oenema *et al* 2009)

The Nitrates Directive (91/676/EEC, adopted in 1991) has been implemented across the EU with each country responding to the requirements in their own way. A recent report by Sutton and Billen (2011) has identified that a more integrated approach will be required to manage the unwanted N emissions from agriculture, combustion and urban wastes in the future.

7.2.1 Water quality in Europe

The inputs of nitrogen (N) to the environment, principally from agriculture, increased by 20% between 1970 and 2010. Emissions per unit area of agricultural land increased by 20–30% due to intensification and peaked in 1985. The legacy of this period of agricultural intensification is a Nitrogen Use Efficiency (NUE) in agriculture across Europe of 30%, compared to 50% for the global average, and high nitrate concentrations in many EU-27 waters. Groundwater concentrations are currently lower than those reported for surface waters. However, concentrations in many groundwaters are rising due to the lag associated with water movement through aquifers (ref to section 6.2). Eutrophication in the fresh water around Europe is improving, but remains a significant environmental problem (Sutton and Billen, 2011).

In the EU–27, the average reduction in nitrate leaching between 1989 and 2001 was 11% (Oenema *et al.,* 2009). However, over the same period, Denmark reduced nitrate losses by 40%. Even so, Denmark's current leaching per unit area of agricultural land is almost twice that of the UK (Berge and Dijk, 2009).

It is reasonable to expect that improvements in water quality will have slowed down in the last decade as many of the most inefficient agricultural practices of the past have already been improved through various initiatives during the 1990s. Improvements are, however, still being reported with one German Land identifying a reduction in nitrate loss of 5–7% in water supply areas between 2001 and 2007 (Berge and Dijk, 2009).

7.2.2 European Action under the Nitrates Directive

The majority of the EU-27 have adopted partial NVZ designation with only 9 member states designating their whole territory (Oenema *et al.* 2009; Table 7.1.) It is important to note, however, that the spatial targeting of measures within NVZs varies considerably from country to country.

Country	Area NVZ, %ᵃ	Country	Area NVZ, %
Austria	100	Italy	27
Belgium	61	Lithuania	100
Bulgaria	0	Luxembourg	100
Cyprus	–	Latvia	13
Czech	38	Malta	–
Germany	100	Netherlands	100
Denmark	100	Poland	2
Estonia	7	Portugal	10
Spain	21	Romania	0
Finland	100	Sweden	49
France	53	Slovenia	100
Greece	19	Slovakia	38
Hungary	45	United Kingdom	81
Ireland	99		

Table 7.1. Percentage of land area within each EU-27 Member State that is designated as NVZ (Source: Oenema *et al.* 2009**)**

A number of countries target the most effective measures in catchments used for potable water abstraction. Denmark, France, Belgium, Germany and the Netherlands all target measures, which are non-optimal for agricultural production, in source protection zones (Berge and Dijk 2009). The use of sub-optimal fertilisation and cover crops significantly reduces leaching. Detailed monitoring of autumn SMN, to identify accurately crop requirements, has helped some German states to reduce nitrate leaching in these protected areas by 7% between 2001 and 2007 (Berge and Dijk, 2009).

Denmark, Belgium, France, Germany and the Netherlands all use cover crops to reduce nitrate leaching. In Denmark, all farmers must have at least 10% of their land under cover crops. In other countries, the use of cover crops is promoted on high leaching sandy and loam soils in groundwater areas or after high risk crops such as maize. High risk shallow soils, sands and loams are targeted in Germany and Belgium with AP measures designed to reduce leaching to groundwater, whereas in the Netherlands, it is the soil/crop combinations that give leaching in excess of 50 mg/l that are targeted. Crop requirements are refined based on previous fertiliser and crop history in both Denmark and Germany.

The management of nitrogen from livestock manures is common to all NVZ APs. The nitrogen fertiliser value assigned to manures within NVZs is higher in most northern European countries than the UK. Closed periods for the spreading of manures are also longer in Denmark, France and Flanders, but in Germany, closed periods are shorter than in the UK (Table 7.2).

Country	Closed period
France	6.5 months on arable and 4 months on grass
Denmark	7 months on arable and 4 months on grass and winter OSR
Flanders	5.5 months
Netherlands	5 months on sand/loess and 4.5 months on clay and peat
Germany	3 months on arable and 2.5 months on grass

Table 7.2. Closed period rules in northern European countries.

Most northern European countries have similar rules for spreading livestock manures aimed at protecting water features and water supplies and minimising the nuisance to residential properties.

Finally, Denmark is generally considered to have been the most successful country in northern Europe at implementing the Nitrates Directive due to its focus on integrated nutrient planning and its improvements in NUE. This success has been attributed to the close dialogue between policy makers and farmers and the extensive use of integrated training and demonstration informed by targeted research.

A recent report by Sutton and Billen (2011) identified that ultimately, a more integrated approach will be required to manage the unwanted N emissions from agriculture, combustion and urban wastes in the future. A cost benefit analysis in the same report identified that lowering the N input rate from agriculture by 50 kg/ha would provide an “optimal” balance between production and environmental costs. To help achieve this reduction, a major shift towards a low meat diet (63% reduction in meat and eggs) would be required. This is because livestock production is N-inefficient. A substantial proportion of all arable crops are fed to livestock, instead of being consumed directly. The report recognises that the impact of such changes on other pollutants and farm incomes could be profound and would require additional research to inform the evidence base.

> *Major reductions in agricultural N_r emissions will occur only if the extent of agricultural production changes, for example linked to changing human populations or per capita consumption patterns. Such a scenario is examined based on a healthier 'low meat' diet leading to lower N losses. (*Sutton and Billen, 2011)*.*

8 SUMMARY REVIEW OF THE EVIDENCE

The main conclusions that can be drawn from this evidence report are:

- Nitrate concentrations in leachate from arable land are typically well in excess of 50 mg/l.
    - This is true even if the farmer is following best practice and applying the current NVZ AP.
    - Smaller nitrate concentrations at catchment scale occur only where there is sufficient dilution with water from non agricultural land or from extensively farmed grassland;
- Nitrate concentrations in many rivers and groundwaters are and remain high
    - There is little evidence that existing NVZ APs have made a significant impact on observed nitrate concentrations in surface water;
- Nutrient surpluses in England and Wales (both N and P) have fallen on grassland systems and P surpluses have fallen on arable over the past 15 years.
    - These reductions in surplus (and any associated local improvement in water quality) are long term trends, possibly due to the fall in livestock farm incomes, and not attributable in large part to the NVZ AP;
- The NVZ AP rules cause greatest change in behaviour and greatest impact on pollution on farms which use manures.
    - Impacts of the current NVZ AP will tend to be greatest in catchments which have a high density of livestock. It will have little impact on pollution in arable areas without manures.
    - In localised areas, the NVZ AP will have a significant impact. The larger the catchment, the more likely it is that the overall impact will be similar to the national average (i.e. small).
    - Additional measures such as the use of cover crops could reduce nitrate losses further, within arable areas on medium to light soils.

REFERENCES

Applegate, T.J., Potturi, L.P.V. and Angel, R. (2003). Model for estimating poultry manure nutrient excretion: a mass balance approach. Intl. Symp. Animal Ag. Food Proc. Wastes 9, 296-302.

Badr, E-S.A., Tappin, A. and Achterberg, E.P.(2008). Distributions and seasonal variability of dissolved organic nitrogen in two estuaries in SW England. Marine Chemistry 110, 153-164.

Berge H.F.M. den and Dijk, W. van (2009). Management of nitrogen inputs on farm within the EU regulatory framework. Proceedings of the International Fertiliser Society No 654. International Fertiliser Society, Leeds, UK.

Boorman, D., Hollis, J. and Lilly, A. 1995. Hydrology of soil types: a hydrologically based classification of the soils of the United Kingdom. *Institute of Hydrology Report No. 126*, Wallingford, Oxfordshire, 134pp

CEH, 1998. Countryside Survey 1998. Centre for Ecology and Hydrology, UK

Chadwick D. R, Matthews R, Nicholson. F, Chambers B.J. & Boyles LO (2002) Management practices to reduce ammonia emissions from pig and cattle manure stores. International Conference of the FAO RAMIRAN Network on Recycling of Agricultural, Municipal and Industrial Residues in Agriculture, General theme, Hygiene Safety - RAMIRAN 2002 (Eds.J. Venglovsky and G. Greserova), pp 219-222.

Chambers, B.J., Smith, K.A. & Pain, B.F. 2000. Strategies to encourage better use of nitrogen in animal manures. *Soil Use and Management* **16**, 157-161

Clothier, L. (2010). Soil Nutrient Balances: 2010 update. www.defra.gov.uk/statistics/files/defra-stats-foodfarm-environ-obs-research-soilnutrientbalance-1108-update.pdf

Cottrill B.R., Sylvester-Bradley, R., Webb, J., Twining, S., Temple, M., Longland, A., Crompton, L., and Mills, J. (2005). A theoretical analysis of how the protein and phosphorus requirements of livestock may best be met. Final report, Defra project IS0208

DEFRA (2011a). *Fertiliser Manual (RB209).* 8th Edition updated. The Stationery Office, Norwich.

Defra (2007). Farm Practice Survey 2007.

Edmunds, W.M., Shand, P., Hart, P. and Ward, R.S. (2003). The natural (baseline) quality of groundwater: a UK pilot study. Science of the Total Environment 310, 25-35.

Edwards A., Sinclair, A. and Domburg. P. (2003). Identification, designation and formulation of an action plan for a nitrate vulnerable zone: a case study of the Ythan catchment, NE Scotland European Journal of Agronomy Volume 20, Issues 1-2, Pages 165-172

Jarvis, S. and Pain, B. (1994). Greenhouse gas emissions from intensive livestock systems: Their estimation and technologies for reduction. Climate change. Vol 27.Number 1 27 - 38

Johnson, D., Fletcher, S.W. and Bray, K. (2007). Integrated assessment of nitrate concentrations for the designation of groundwater Nitrate Vulnerable Zones in England and Wales, Quarterly Journal of Engineering Geology & Hydrogeology; November 2007; v. 40; no. 4; p. 407-415.

Johnson, D. and Lord, E. (2011). Assessment of trends in N and P balance in Agriculture, by sector, with likely causes. EPCRS Farmework project 14627 for Environment Agency.

Jones, D.L., Shannon, D., Murphy, D.V. and Farrar, J. (2004). Role of dissolved organic nitrogen (DON) in soil N cycling in grassland soils. Soil Biology and Biochemistry, 36, 749-756.

Kay, D., Anthony, S.G., Crowther, J., Chambers, B.C., Nicholson, F.A., Chadwick, D., Stapleton, C.M. and Wyer, M.D. (2010). Microbial water pollution: a screening tool for initial catchment-scale assessment and source apportionment. Science of the Total Environment, 408, 5649-5656.

Lord, E., Johnson, P. and Archer, J. (1999) Nitrate Sensitive Areas – a study of large-scale control of nitrate loss in England. Soil use and Management 15, 1-7

Lord, E. and Mitchell, R. (1998) Effect of nitrogen inputs to cereals on nitrate leaching from sandy soils. Soil Use and Management, 14, 78-83

Lord E.I., Cottrill, B., Newell-Price, P. and Smith, K. (2010). UK Nutrient Balances: Methodology review. Final report on Defra/FERA project T4QT

Nicholson. F., Chambers. B., and Williams. J. (in press) WT0932. Closed period report. Defra.

Murphy, D.V., Macdonald, A.J., Stockdale E.A., Goulding, K.W.T., Fortune, S., Gaunt, J.L., Poulton, P.R., Wakefield, J.A., Webster, C.P. and Wilmer, W.S. (2000). Review article: Soluble organic nitrogen in agricultural soils. Biology and Fertility of Soils 30, 374-387.

Oenema ,O, Witzke, H.P., Klimont, Z, Lesschen, J. and Velthof, G. (2009). Integrated assessment of promising measures to decrease nitrogen losses from agriculture in EU-27 Agriculture, Ecosystems & Environment Volume 133, Issues 3-4, October 2009, Pages 280-288

Scholefield, D., Lockyer, D., Whitehead, D. and Tyson, K. (1991). A model to predict transformations and losses of nitrogen in UK pastures grazed by beef cattle. Plant and Soil 123, 165-177.

Shepherd, M. (1999). The effectiveness of cover crops during eight years of an UK sandland rotation. Soil Use and Management 15, 41-48.

Siemens, J. and Kaupenjohann, M. (2002). Contribution of dissolved organic nitrogen to N leaching from four German agricultural soils. Journal of Plant Nutrition and Soil Science 165, 675-681.

Sommer S.G. and Hutchings N.J. (2001). Ammonia emission from field applied manure and its reduction – invited paper. European Journal of Agronomy 15, 1-15.

Stapleton, C.M., Kay, D., Magill, S.H., Wyer, M.D., Davies, C., Watkins, J., Kay, C., McDonald, A.T. and Crowther, J. (2011). Quantitative microbial source apportionment as a tool in aiding the identification of microbial risk factors in shellfish harvesting waters: the Loch Etive case study. Aquaculture Research 42, Supplement S1, 1-20.

Stopes, C., Lord, E., Philipps, L. & Woodward, L. (2002) Nitrate leaching from organic farms and conventional farms following best practice. Soil Use and Management 18, 256-263

Sutton. A and Billen. G. (2011). The European Nitrogen Assessment: Sources, Effects and Policy Perspectives. Cambridge, Cambridge University Press, xxxv-li

Williams AT, Hughes AG and Chilton JC. 2011. Nitrates in groundwater: An explanation and description of response times in groundwater for the Article 10 reporting process. *British Geological Survey Internal Report*, OR/08/043. 65pp

Williams J.R. and Nicholson, R.J. (1995). Low rate irrigation of dirty water and its effect on drainage quality. Agric Eng. 50, 24-28.

Worrall., F., Spencer, E. and Burt, T.P. (2009). The effectiveness of nitrate vulnerable zones for limiting surface water nitrate concentrations. Journal of Hydrology 370, 21-28.

Annex 1: Questions from the 'Consultation on Implementation of the Nitrates Directive in England' and the supporting evidence that can be found in this document.

Question 1: Do you prefer Option 1 (continuing with discrete NVZ designations) or Option 2 (applying the Action Programme to a 'Whole England' NVZ)?

- Section 7.2.2. Designation practice in the rest of Europe.

Question 3: Do you agree that crop available nitrogen from all types of organic manures should count towards the Nmax limits?

- Section 5.2. shows the saving in manufactured fertiliser due to the increasing use of the nutrient content of manures.

Question 8: Which of the 3 closed spreading period options do you prefer?

- Section 3.3.2. The impact of soil type on nutrient losses.
- Section 3.4.1. Manure timing and 'Closed Spreading period'
- Section 3.12. The importance of the timing of nutrient losses on ecology.
- Section 4.1. The timing of manure applications and the impact on losses of phosphorous and sediments.
- Section 4.4 Ammonia losses from manures
- Section 7.2.2. Closed periods in other European countries.

Question 9: Do you support the above closed spreading period based on rainfall banding?

- Section 3.3.1 The influence of rainfall on nutrient losses.

Question 10: Do you think that reducing the quantity of slurry and other materials that can be spread during and immediately after the closed period is a better mechanism for managing nitrate leaching than extending the closed periods?

- Sections identified for question 8.

Question 11: Do you agree with the proposals to reduce the minimum distance for spreading slurry near watercourses if a precision slurry spreader is used?

- Section 4.3. Pollutant losses
- Section 7.2.2. Most European countries have similar rules to England on the spreading of slurry.

Question 13: Do you agree that the Action Programme does not require any amendments with respect to the storage of solid livestock manures?

- Section 3.9. Field Heaps

Question 16: Do you think cover crops should be included in the Action Programme?

- Section 3.7. Winter cover crops
- Section 6.2.1. Spatial impacts of cover crops
- Section 7.2.2. What the rest of Europe does