INVESTIGATING ASSESSMENT METHODS FOR THE EVALUATION OF ACTIONS MITIGATING NITRATE LOSS TO WATER

KATHERINE A. CHERRY, BSc. (Hons.)

Thesis submitted to the University of Nottingham for the degree of Doctor of Philosophy

JULY 2011

Abstract

Diffuse nitrate (N) loss from agriculture is degrading surface and groundwater quality throughout Europe, leaving waterbodies at risk of not reaching targets set by the Water Framework Directive (WFD). Although a wide range of mitigation methods to reduce diffuse N loss have been identified, their appropriateness and effectiveness is not fully understood, especially at the catchment scale where a wide range of environmental and agricultural conditions exist. Suitable assessment methods are required to quantify the impact of mitigation and provide confirmation of their effectiveness. This study aimed to investigate the applicability of measurement and nutrient budgets for the evaluation of mitigation effectiveness at the field, farm and catchment scale; nutrient budgets represent an alternative approach where long transit times delay observable responses to mitigation in measurement. Investigations focused on two catchments in SW England, Milborne St Andrew (MSA) and Empool / Eagle Lodge (EMEL). Soil surface budgets were calculated for a total of 84 fields and farmgate surpluses / efficiency for 34 farms between 2005 and 2008. Soil mineral nitrogen (SMN) and porous pot (PP) sampling was undertaken in 115 and 57 fields respectively, and groundwater / stream water monitored at 171 sites. Sampling was carried out in 2007 and 2008, and a range of mitigation methods adopted on farm in 2008. Comparing results before and after mitigation, measurement approaches displayed contradictory responses – SMN significantly decreased, PP leached load and concentration significantly increased, and groundwater responses varied between sites. Results suggest an overriding sensitivity to environmental condition and the need for longer timescales especially at the catchment scale. Nutrient budgets at the field and farm scale tended to return lower surpluses post mitigation with 79% / 77% farms improving their farmgate surplus / efficiency. However only in EMEL were improvements in field or farm scale surpluses significant, a result of modest mitigation induced change and sensitivity to economic and environmental drivers. Comparing measurement and budget approaches, budgets were more responsive to changes in nutrient management in the short term and offered higher levels of farmer accountability. However long term measurements are required to provide confirmation that improvements in nutrient budgets transpire in water quality. As such a combined approach is suggested. With direct links to economic benefits likely to aid farmer engagement, and providing more complete representations of mitigation response and feedback, the use of farm scale budgets / efficiency over field scale budgets is advocated.

Acknowledgements

I would like to thank my (many!) supervisors, Dr Sacha Mooney, Prof. Mark Shepherd, Dr Paul Withers, Dr Stephen Ramsden and Dr Jodie Whitehead for their guidance and patience over the last 4 years, and for coping with my at times rather particular ways of working - I have just about accepted that the perfect thesis might not exist! I also gratefully acknowledge funding from the University of Nottingham School of Biosciences and from ADAS.

My thanks must also extent to the hardy ‘WAgriCo Workers’ (Matt Taylor, Tony Lloyd and Geoff Bailey) for their invaluable practical assistance on cold / wet / dark days SMN-ing and porous potting. As enjoyable as those days were, I will not miss the sweet smell of my pick up, the temperamental hydrocare or crawling across fields to avoid being spotted by cows!

The last four years would not have been the same without the lovely people I have met along the way and all the fun that has been had .... ceroc-ing in Salisbury, faffing and de-caffing with the girls at Woodside Road and Dennis Avenue, dinner (not lunch) time with B99-ers, and shower time with the speedy Salisbury Stingrays, Bramcote-ers and STAM-ers.

During the seemingly endless days of writing up, thanks Lees for ‘chivvying’ me along with our mutual target setting and progress reports, Nick for our amazing voucher fun, and Lisa for ensuring I had a plentiful supply of recycling / garden ‘opportunities’ which made writing not seem so bad after all! I did very much appreciate having a roof over my head though!

And last but by no means least, I would like to thank my wonderful Mum and Dad for all their encouragement, love, support and funds (!) ... although sorry Dad I didn’t quite get round to the chapter on climate change! And to my very special sisters Lisa and Villa, thanks for the ‘Cherry Support’. It seems I might have lived up to my ‘slow but steady wins the race’ motto once again!

Contents

List of Tables

List of Figures

Abbreviations

C	Carbon
CP	Crude Protein
DM	Dry Matter
DWS	Drinking Water Standard
EC	European Commission
EEA	European Environment Agency
EGAP	Enhanced Good Agricultural Practice
EMEL	Empool / Eagle Lodge
EU	European Union
EU-15	EU member states prior to 2004
EU-25	EU member states prior to 2007
EU-27	EU member states from 2007
FW	Fresh Weight
FYM	Farm yard manure
GAP	Good Agricultural Practice
Ha	Hectare
LU	Livestock Units
mAOD	Metres Above Ordnance Datum
MMP	Manure Management Plan
MSA	Milborne St Andrew
N	Nitrogen
NO_3-N	Nitrate as N
NH_3	Ammonia
NH_4^+	Ammonium
NSA	Nitrate Sensitive Area
NVZ	Nitrate Vulnerable Zone
PofMs	Programme of Measures
PP	Porous Pots
RAN	Readily Available N
RBMP	River Basin Management Plan
SBF	Spring barley (feed varieties)
SBM	Spring barley (malting varieties)
SE	Standard Error
SMN	Soil Mineral Nitrogen

TAN	Total Ammoniacal Nitrogen
TN	Total Nitrogen
WAgriCo	Water Resources Management in Co-operation with Agriculture
WBF	Winter barley (feed varieties)
WFD	Water Framework Directive
WO	Winter oats
WOSR	Winter oilseed rape
WWF	Winter wheat (feed varieties)
WWM	Winter wheat (milling varieties)

1 Introduction

1.1 Project Rationale

Nitrate-N (NO_3-N, but referred to from here on in as N) enrichment is degrading water quality throughout Europe, inducing eutrophication and resulting in non-compliance with legislation. Nitrate concentrations exceed the Drinking Water Standard (DWS) of 11.3mg N l^{-1} in 15% of groundwater bodies and 3% of surface water bodies in EU-27 member states (EC, 2010a). Concentrations in UK waterbodies are high compared to other EU member states with 32% of rivers in England and Wales classified as having high nitrate concentrations (> 6.8mg N l^{-1}, corresponding approximately with the 95th percentile of the 11.3mg N l^{-1} limit) (Environment Agency, 2008). In an attempt to protect and enhance aquatic ecosystems, the Water Framework Directive (WFD) (2000/60/EC) was introduced in 2000. All water bodies in European Union (EU) member states are required to reach 'good' and 'non-deteriorating' status by 2015, alike to the conditions observed under minimal anthropogenic influence. Surface waters must achieve good ecological and chemical status, whilst groundwaters must reach good chemical standard and pose no risk to the status of surface water into which they may flow. At present, 60% of groundwater in England and Wales is at risk of failing to meet WFD targets due to high N concentrations (Environment Agency, 2006).

Existing / previous legislation, namely the Urban Waste Water Treatment Directive (91/271/EEC), has been effective in reducing large urban point sources of N. In most catchments N loads are now predominately diffuse i.e. losses are widespread and their exact origin difficult to identify. Agriculture represents a significant source of N, contributing 61% of annual loads to water in England and Wales (Hunt et al., 2004), figures typical of those reported throughout Europe (EEA, 2005). Nitrate is highly soluble and readily leached through the soil. Loss is attributed to many factors, including over fertilisation (Lord and Mitchell, 1998), excessive manure applications, a failure to consider the nutrient content of manure in fertiliser recommendations, poorly timed nutrient applications (Smith et al., 2001), autumn ploughing, and intensive stocking of pasture (Shepherd et al., 2001; Shepherd and Chambers, 2007). Management practices interact with the inherent variation in soil type, climate, topography, geology and hydrology giving rise to large spatial and temporal variation in nutrient concentrations in land runoff. The scattered distribution of diffuse sources and complex delivery processes makes targeting these sources difficult (Heathwaite et al., 2005a).

Diffuse agricultural pollution must be controlled in order to minimise the adverse impacts of agriculture on ecosystems and to comply with legislative requirements. An Integrated Catchment Management (ICM) approach has been adopted in England and Wales whereby land and water are managed together (Burt, 2001); the WFD is driving this approach throughout Europe. Water bodies at risk of not reaching ‘good’ status are being identified and programmes of measure (PofMs) devised to increase their likelihood of meeting WFD targets. In response to the WFD and to tackle diffuse nutrient pollution internationally, a wide range of ‘mitigation methods’ have been developed that can be adopted on farm to reduce diffuse agricultural nutrient loss. Mitigation methods affect soil, livestock, manure and fertiliser management, and farm infrastructure (Cuttle et al., 2006), targeting nutrient availability, the timing of agricultural practices and the delivery of nutrients from source to recipient water bodies.

Commitment to water quality and ecological standards under the WFD has generated a need to assess the effectiveness of mitigation and quantify impact across a range of scales. Evaluations provide assurance that PofMs are making progress towards the goals of the WFD whilst confirmation of success under specific conditions ensures implementation is increasingly targeted and thus more cost effective. Mitigation options have been evaluated at field and farm scale (e.g. Johnson and Smith, 1996; Shepherd, 1999; Johnson et al., 1997 and 2002), however most assessments have been performed using designed field experiments (e.g. Beckwith et al., 1998) or derived from modelled simulations of typical farm scenarios (e.g. Cuttle et al., 2004; Cuttle et al., 2006). Despite being more relevant to the waterbody focus of the WFD, fewer evaluations have been conducted at the catchment scale because of the difficulty in covering a wide range of environmental and agricultural conditions. As a result there remains a need to assess the impact of mitigation adopted in a practical context especially at the catchment scale. The implementation of mitigation within two Dorset catchments as part of the EU Life funded WAgriCo project provided an opportunity to assess impact on measured losses, quantifying impact beyond that on modelled farms or under experimental conditions.

The quantification of mitigation effectiveness requires suitable methods of assessment. Measurement represents the most direct means of asesssing mitigation effectiveness, quantifying changes in water quality / N loadings following the implementation of mitigation. Assessments of mitigation will, in the first instance, be performed using field / catchment measurement approaches. However its interpretation is confounded by sensitivity to environmental factors, with annual variability concealing responses to mitigation. Superimposed upon this is the issue of timelags which delay observable responses to mitigation especially at the catchment scale. With good status required by 2015 and the success of PofMs reported on a six yearly basis, assessment methods capturing the impact of mitigation in the short term are of particular relevance. As such there is a need to assess the usefulness of measurement for the evaluation of mitigation effectiveness in the short term. Previous short term assessments have focussed on the impact of mitigation and not the suitability of the assessment approach. While catchment scale timelags typically exceed the four year timescale available, responses in some catchments are seen much sooner. Given the integrated nature of catchment scale responses and greater relevance to waterbody status, catchment responses are preferable. Investigations into the usefulness of short term measurement should be extended to the catchment scale to determine whether in some locations catchment scale measurement represents an effective short term assessment method.

Despite their continued use, long timescales and difficulty interpreting results means measurement-only based evaluations are costly and uncertain. This raises the question of whether alternatives exist and how their performance compares to measurement based approaches. To date this has not been directly or quantitatively addressed. The literature points to nutrient accounting as a potential alternative, in which inputs and outputs are evaluated to determine whether surpluses or deficits exist. With surplus nutrients at risk of loss to the environment, a reduction in the surplus is likely to yield environmental benefits. Mitigation effectiveness can therefore be quantified by the magnitude of improvements in nutrient surpluses following its implementation. Closely linked to nutrient budgets is the concept of efficiency. The ratio of useful product to input is calculated, with inefficiencies reflecting N at risk of loss. Providing opportunities to conduct more detailed evaluations, respect production intensity, and offer more communicable results, efficiency represents another candidate approach. The usefulness of a range of nutrient budget and efficiency approaches should be explored and compared to measurement approaches. Investigations should aim to establish whether budget /

efficiency approaches represent an effective alternative to measurement based assessment. This will in part require exploration of the relationship between surpluses and measured loss to determine the extent to which improvements observed in surpluses translate to reductions in measured loss.

The scale dependency of nutrient loss processes (Quinn, 2004) means apparent mitigation effectiveness differs between field, farm and catchment scale. Coupled with the WFD being implemented and assessed at the catchment scale, scale is an important consideration with regard methods of assessment. Despite considerable interest in the effect of scale on losses and the limits of extrapolation, the effect of scale on assessments of mitigation effectiveness has received little attention. As a result assessment methods should be applied at field, farm and catchment scale to investigate the effect of scale on current / alternative assessment methods. Scale applicability represents one of a number of assessment method traits with communicability, uncertainties, and legislative relevance also requiring consideration. Comprehensive assessments and comparisons of current and alternative assessment methods should culminate in the recommendation of most suitable approaches under specific circumstances at field, farm and catchment scale.

1.2 Aims and objectives

1.2.1 Overall aim

To investigate the applicability of measurement and nutrient budget based approaches for the evaluation of mitigation effectiveness at field, farm and catchment scale.

1.2.2 Objectives and hypotheses

1. To assess the usefulness of measurement (field / farm and catchment) for the short term evaluation of mitigation effectiveness.
    - *‘Field and catchment scale measurement represent effective methods of mitigation evaluation’ – Chapter 4*

2. To identify and assess the usefulness of field and farm scale budget based assessment approaches

    - *'Field scale nutrient budgets represent an effective method of mitigation evaluation.'* – Chapter 5

    - *'Farm scale nutrient budgets represent an effective method of mitigation evaluation.'* – Chapter 6

    - *'Efficiency represents an effective method of mitigation evaluation'* – Chapter 7

3. To investigate the effect of scale on measurement and budget based assessments of mitigation effectiveness.

    Field vs. catchment measurement – Chapter 4

    Field vs. catchment surpluses (soil surface approach) – Chapter 5

    Farm vs. catchment surpluses (farmgate approach) – Chapter 6

4. To investigate links between nutrient budget and measurement approaches for the support of budget based approaches – Chapter 8

5. To compare the usefulness of measurements and budget based approaches – Chapter 8

6. To make recommendations regarding the most suitable approach under specific circumstances at field, farm and catchment scale – Chapter 8

1.3 Thesis structure

To address the aims and objectives of this study, various measurement and nutrient budget based approaches were used to assess the impact of mitigation adopted in a practical context in two south Dorset catchments. Details of the catchments and the mitigation implemented are presented in chapter 3.

Investigations into the usefulness of measurement and nutrient budget approaches are presented on a method by method basis (chapters 4 – 7). Widely used measurement approaches were supplemented by budget approaches highlighted in the literature as having potential to provide assessments of mitigation effectiveness. Budget methodologies were developed / refined as necessary to maximise suitability to evaluations of mitigation effectiveness in the study catchments. Assessments of mitigation impact were preceded by assessments of spatial and temporal variability in nutrient use / loss. Nutrient characterisation provided assurance that approaches were robust whilst also offering an insight into method sensitivity should mitigation have had little impact. Sensitivity to mitigation was assessed on an individual mitigation method basis and on a cumulative mitigation basis for each approach (where applicable). Opportunities to upscale results, compare results between scales, perform sensitivity analysis and simulate mitigation scenarios were exploited where possible. Such investigations sought to support and broaden the analyses already conducted and investigate in more detail the effect on scale on evaluations.

Chapter 8 focuses on the interactions between methods, investigating the effect of nutrient budget methodology on results, and the links between nutrient surpluses and loss. Confirmation of links between surpluses and loss supports the use in nutrient budgets, whilst comparison of budget methodologies exposed differences between methods. Chapter 8 compares the approaches investigated, culminating with recommendations as to their suitability to evaluations at field / farm / catchment scale under specific circumstances.

2 Literature Review

(Some of the discussions that follow have been published in Cherry et al., (2008))

2.1 The state of European water bodies

2.1.1 N concentrations in surface and groundwater

Nitrate (N) enrichment is degrading water quality throughout Europe, inducing eutrophication and resulting in non-compliance with legislation. Currently 3% of surface waters in EU 27 member states exceed the EC's Drinking Water Standard (DWS) of 11.3mg N l^{-1} (EC, 2010a). Nitrate concentrations in UK surface waters are higher than those observed in most member states, with the proportion of surface water sites exceeding the DSW increasing to 7% (EC, 2010a). The mean nitrate concentrations in UK rivers is higher than in France, Netherlands and Germany, and of the EU-15 the UK has the largest proportion of rivers in the highest >7.5mg N l^{-1} category (EEA, 2004). As a result 32% of rivers in England and Wales have nitrate concentrations in excess of 6.8mg N l^{-1} (corresponding approximately with the 95th percentile of DWS) (Environment Agency, 2008). While the picture across the rest of Europe is generally better than in the UK, high nitrate concentrations have also been observed in surface waters in Denmark, Belgium and parts of France (EC, 2010a).

Nitrate is also degrading groundwater with a third of EU-27 groundwater bodies exceeding guideline nitrate concentrations (5.7mg N l^{-1}) and 15% above the DWS (EC, 2010a). In 2005 19 out of 31 European countries had waterbodies with concentrations above the DWS (EEA, 2009). In Western Europe mean groundwater concentrations are above the 5.7 mg N l^{-1} guideline; this is in contrast to Nordic countries where concentrations are consistently low (EEA, 2004). As observed in surface waters, nitrate concentration in UK groundwater is high relative to the rest of Europe. Ranked according to the proportion of monitoring sites exceeding the DWS, the UK is in the highest third (EEA, 2004). However concentrations in excess of 9mg N l^{-1} are also observed in Estonia, Belgium, Malta, Cyprus and parts of France, Netherlands, Italy, Spain, Slovakia and Romania (EC, 2010a).

Concentrations of nitrate in surface waters show considerable regional variability corresponding with differences in land use, soil type and climate. Highest concentrations are observed in central, eastern and southern parts of the UK. For example 64% of rivers in the Environment Agencies Anglian region exceeded concentrations of 6.8mg N l^{-1} in 2008 (corresponding approximately with the 95th percentile of DWS). In contrast only 10% of rivers in the Northwest region recorded similar concentrations (Environment Agency, 2008).

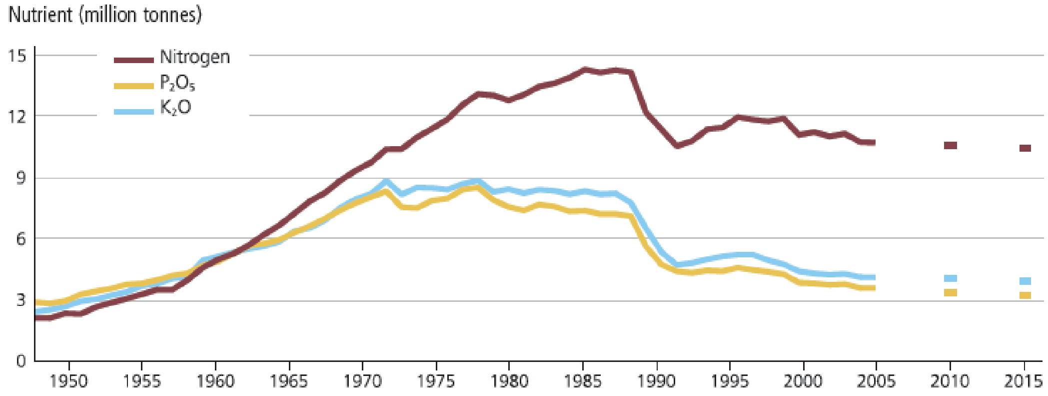


Figure 2-1: Fertiliser consumption (million tonnes) in EU-25 countries between 1948 and 2015 (projected figures between 2005 and 2015) (European Fertiliser Management Association, 2010).

2.1.2 Trends in N concentration

In many European countries, nitrate concentrations in surface waters increased between 1950 and 1990 corresponding with an increase in the use of chemical fertiliser (Kristensen and Hansen, 1994) (Figure 2-1). 70% of European rivers experienced an increase in concentrations between 1978-1988 and 1988-1990 (Kristensen and Hansen, 1994). Increases were most pronounced in eastern and southern European countries where fertiliser use peaked later than in north-west Europe (EEA, 2004). Between 1990 and 1998 nitrate concentrations in rivers across Europe showed little change with concentrations remaining stable across all river types (EEA, 2004). However between 1992 and 2005 35% rivers displayed a significant decreasing trend in nitrate concentration with only 3% displaying a significant increase (EEA, 2009); average nitrate concentration across EU-27 countries reveal a corresponding decrease (Figure 2-2). This improving trend appears to have continued. The EC recently reported that 70% of surface waters in EU-15 countries remained stable or decreased between current (2004-2007) and previous (2000-2003) monitoring periods. It was suggested that improvements reflect a positive response to legislation concerning agricultural nutrient use, namely the Nitrate Directive (91/676/EEC) – see section 2.4.3 for more details. However improvements were not observed EU wide with upward trends in France and Sweden (EC, 2010a).

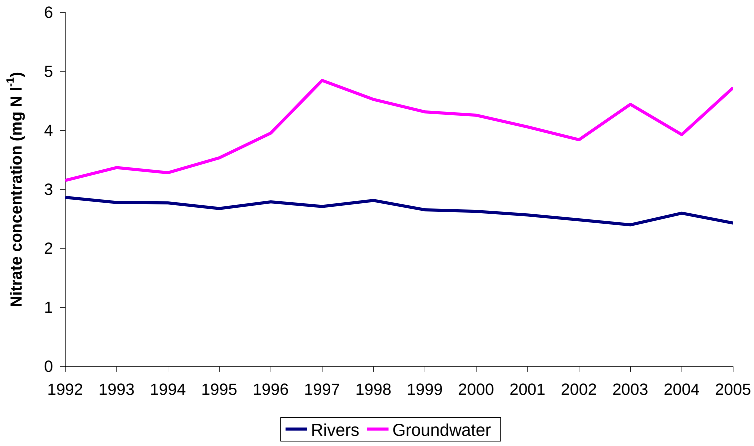


Figure 2-2: Annual mean nitrate concentrations (mg N l^{-1}) in rivers and groundwater across EU-27 countries (EEA, 2009)

The situation is less positive in the UK where the average river nitrate concentration has not fallen since 1980, instead displaying a significant increase of 0.02mg N l^{-1} per year (Worrall et al., 2009). In addition the flux of nitrate to coastal waters has also increased (Worrall et al., 2009). In recent years the proportion of monitoring sites exceeding the DWS has also increased, however there is considerable regional variability with a large proportion of sites in Western England displaying a reduction in concentration (EC, 2010a). However data from the Environment Agency paints a slightly different picture; differences in reporting methods may limit the validity of direct comparison. While the percentage of rivers in England exceeding the 6.8mg N l^{-1} threshold increased between 1990 and 2000 from 36 to 39%, this has since declined to 32% (Figure 2-3). In contrast to the EC's report, maximum improvements have been observed in the Midlands and east of England. While concentrations in rivers flowing through predominately lowland pasture are half those under arable production, an upland trend is particularly evident in the latter.

Over the last 30 years increases in groundwater nitrate concentration were also observed across Europe. Trends in nitrate concentrations between 1992-1994 and 1996-1998 for example were summarised by the European Environment Agency (EEA, 2004) as being stable to increasing and the average nitrate concentration across all EU-27 sample sites increased until 1997 (Figure 2-2). However since the mid 1990's nitrate concentrations have stabilised, and in a large proportion of monitoring sites have begun to improve (EEA, 2009; EC, 2010a; Figure 2-2). The

European Commission reported that 32% of groundwater monitoring sites displayed a significant decreasing trend between 1992 and 2005. However an increasing trend was still observed at 11% of sites (EC, 2010a). The proportion of sites improving has since increased with concentrations falling or remaining stable at 66% of groundwater sites between 2000-2003 and 2004-2007 (EC, 2010a). However in the UK upward trends continue to be observed in around a third of sites with the number of sites exceeding the DWS also increasing (EC, 2010a).

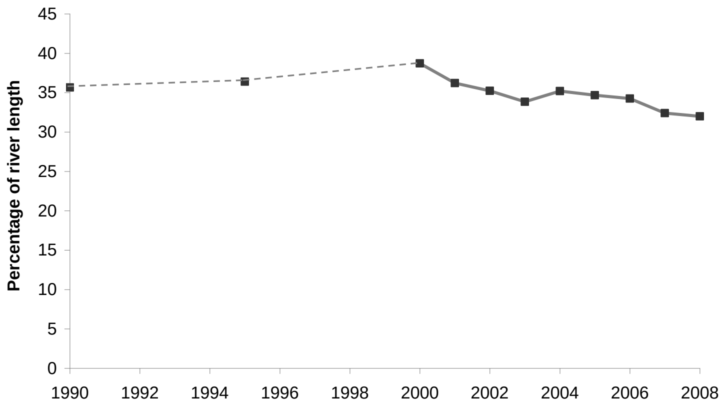


Figure 2-3: Percentage of English river length with N concentrations >6.8mg N l^{-1} between 1990 and 2008 (Environment Agency, 2008).

2.2 Agriculture and N loss

2.2.1 Sources of N

Nitrate (N) in ground and surface waters originate from a variety of sources including agriculture, sewage works, industrial discharge, atmospheric deposition and non agricultural land uses such as woodland and rough grazing land. Losses from agricultural land represent the largest fraction of loadings, estimated to account for 61% of N which enters surface waters in England and Wales (Hunt et al., 2004). Similar figures are reported throughout Europe with the EEA concluding that 50-80% of total N loadings to surface waters originate from agriculture (EEA, 2005). Sewage works account for a further 30% of inputs in England and Wales, however the relative magnitude of both agricultural and sewage inputs varies between regions reflecting the prevalence of agricultural and population density. Agricultural inputs are largest in the South West of England and least in Southern and Thames regions (Hunt et al., 2004). In contrast to inputs from industrial and sewage treatment works, N loss from agricultural land cannot be attributed to a specific location and is referred to as being diffuse.

2.2.2 N loss from agriculture

Nitrogen occurs naturally in the environment cycling between the atmosphere, soil, water and plants, however this system is modified by agricultural production (Figure2-4). Nitrogen is essential for the synthesis of proteins and hence for the production of crops and livestock. Nitrogen is applied to crops as fertiliser to replenish nutrients and maximise yields whilst feed crops and concentrates supply N to livestock which is later returned to the soil as manures. The addition of fertiliser and manure to the soil supplements pools of inorganic (NH_4^+ / NO_3^-) and organic N (R-NH_2) occurring naturally within the soil, increasing the supply of crop available N. Fertiliser is applied as nitrate (NO_3^-) and / or ammonium (NH_4^+) to maximise crop availability; nitrate can be taken up by crops directly while ammonium is rapidly nitrified to nitrate. Manures supply N primarily in organic forms requiring mineralisation and nitrification before being crop available. The incorporation of crop residues also returns organic matter to the soil whilst leguminous crops 'fix' atmospheric nitrogen gas from the atmosphere via symbiotic relationships with micro-organisms.

Nitrogen is lost from all soils, however agriculture increases the likelihood and magnitude of loss. Where the supply of N exceeds crop demand nitrate is at risk of loss, and where manures are stored or spread ammonia volatilisation occurs. Nitrate loss depends on both the availability of N and the effectiveness with which transport processes facilitate its loss; nitrate is highly soluble and poorly adsorbed to soil surfaces facilitating loss via leaching should drainage occur. Nitrate losses from agricultural soils are therefore governed by a range of farm management and environmental factors which are discussed below.

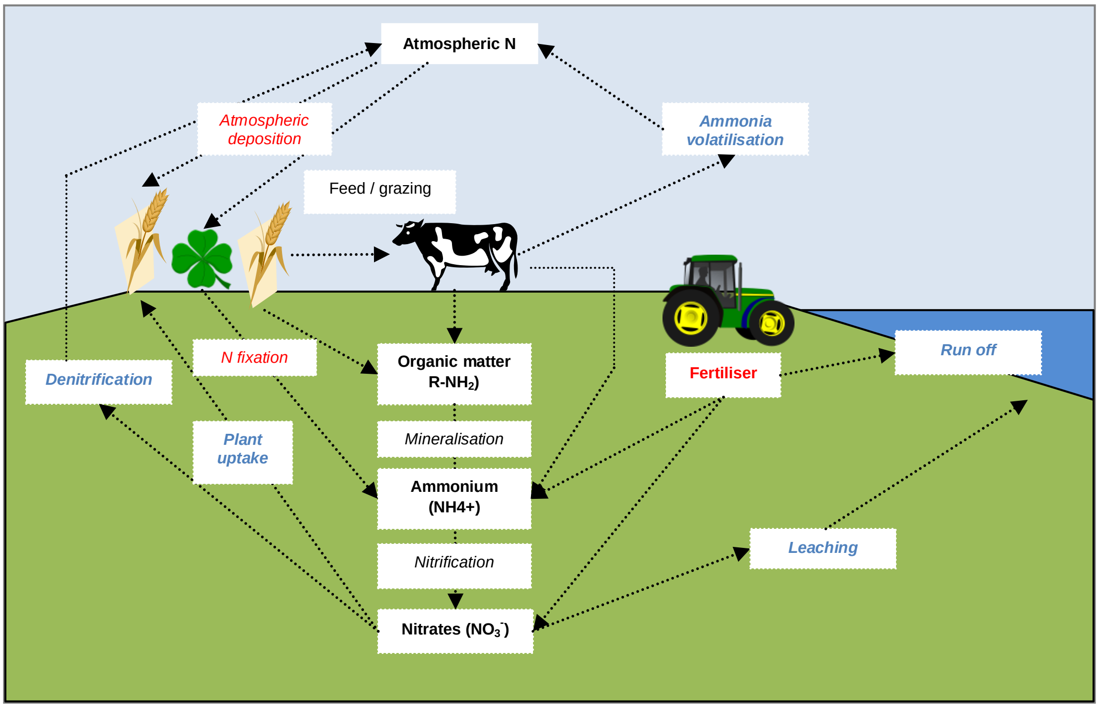


Figure 2-4: The effect of agriculture on the N cycle.

2.2.2.1 Fertiliser inputs

Nitrate loss increases with increasing fertiliser inputs, a result of inefficient crop uptake. However the relationship is not constant across all levels of N inputs with the proportion of N inputs lost considerably higher above the economic optimum. For every 1kg of fertiliser N applied per ha below this break point around 0.05kg N ha^{-1} is lost compared to 0.52kg N ha^{-1} above. Inputs are more effectively converted into crop N at lower inputs, hence the economic optimum (Lord and Mitchell, 1998). Above this it is not profitable to apply additional fertiliser because the yield response does not match the additional cost of fertiliser. So while the application of fertiliser increases the risk of loss, inputs above the economic optimum cause a disproportionate amount of N loss. The timing of fertiliser applications also affects the likelihood of loss. Where supply coincides with demand a larger proportion of available N will be utilised by the crop leaving less at risk of loss.

2.2.2.2 Manure applications

The addition of manures to arable and grassland increases the size of organic and inorganic (readily available and leachable) N pools. Inorganic fractions provide crop available N in the short term, whilst organic matter is mineralised releasing additional ammonium / nitrate in the medium to long term. While the receipt of manure does not increase the risk of loss per say, applications are often poorly timed, excessively large and not accounted for in fertiliser planning increasing the risk of loss (Smith et al., 2001). Beckwith et al., (1998) observed highest leached losses following manure applications in September, October and November whilst losses following applications in December and January were not significantly higher than from untreated controls. Where manures are applied earlier, crop uptake is low and available N poorly utilised. With respect to application rate Lord et al., (1999) reported average losses of 51, 88, 99 and 455 where 0, <175, 175-350 and >350kg N ha^{-1} was applied as manure to cereal crops. With manure often treated as a waste product and not a valuable nutrient rich resource, applications are often too large, especially when applied in addition to fertiliser, to be utilised by the growing crop. The likelihood of loss is also affected by the type of manure applied with lower losses associated with farm yard manure (FYM) than broiler litter and slurry (Beckwith et al., 1998). This is a result of differences in the proportion of total N in soluble and rapidly mineralisable N forms.

2.2.2.3 Land management

The cultivation of soils, especially those that are poorly structured increases mineralisation and the risk of N loss (e.g. Hansen and Djurhuus, 1997); turnover of the soil increases soil aeration and temperature, and brings soil micro-organisms into contact with fresh, previously unavailable substrates (Silgram and Shepherd, 1999). Losses are also affected by the level of cultivation with larger losses associated with ploughed than shallow cultivated fields (Johnson et al., 1997; Salo and Turtola, 2006). In addition losses are typically higher where cultivation occurs in the autumn rather than in the spring (Hansen and Djurhuus, 1997), and where autumn cultivation takes place earlier in the autumn (Johnson et al., 1997). Cultivation also affects the development of flow pathways. Regular cultivation disrupts the development of macropores, thereby reducing the likelihood of preferential flows in poorly structured soils (Hansen and Djurhuus, 1997).

Crop residues are chopped, baled and removed from the field, or incorporated during cultivation. The incorporation of residues supplies additional organic matter which is mineralised and available for loss or uptake by the following crop. The quantity of N returned to the soil depends on the preceding crop; WOSR for example leaves large residues (Goulding, 2000). However the incorporation of straw affects the ratio of C:N with the potential to immobilise N and reduce the risk of leaching as demonstrated by Silgram and Chambers (2002). In both cases the impact of residue incorporation is affected by timing. Owing to higher mineralisation and increased potential for leaching prior to the establishment of the next crop, the risk of leaching is higher where residues are incorporated earlier especially when drainage begins early in the autumn (Mitchell et al., 2000).

However the affect of cultivation date on loss is intertwined with the date of drilling. Losses are maximised where the lag between the two are high during which significant mineralisation can occur and uptake is low. Accordingly Johnson et al., (1997) observed high losses from peas and oilseed rape where ploughing was early and the next crop (in this case peas and oilseed rape) sown late. Losses were much lower where wheat was drilled early. These findings support Stokes et al., (1992) and Catt et al., (1992) who found that delaying primary cultivation reduces mineralisation and can decrease N loss where crops are successfully established soon after. In the absence of crop cover available N is at risk of loss.

2.2.2.4 Land use and cropping

Crop type and land use (arable vs. grassland) have a substantial affect on nutrient management and land management. The likelihood of nitrate loss differs between crops. Maximum losses are associated with potatoes, peas and rotational set-aside (Goulding, 2000). Potatoes and peas leave large residues whilst negligible crop uptake and cultivation induced mineralisation elevates losses from rotational set-aside (Lord et al., 2007). Nutrient requirements are crop specific, as are their ability to capture and utilise them efficiently. Fertiliser inputs to winter oilseed rape and winter wheat are high, but while wheat captures N less efficiently, that which it does absorb is more efficiently converted into useful product. Fertiliser inputs to potatoes and sugar beet are typically lower but uptake efficiency and offtake both high (Sylvester Bradley and Kindred, 2009). Grown as livestock fodder, maize is often integrated within grassland systems and more likely to receive manure.

Losses from grass reflect the intensity of the livestock production system, increasing with increasing fertiliser inputs and stocking rates (Lord et al., 2007). However the continual crop cover and nutrient uptake associated with grass and the lack of cultivation (reducing mineralisation inputs) means losses are lower than levels of inputs might suggest (Lord et al., 2007). Grass fields are likely to receive manures and / or be grazed. Concentrated inputs of N in ‘urine patches’ can disproportionally increase losses associated with excretal return during grazing compared to that of manure spreading (Cuttle et al., 2001). However where stocking rates are high relative to farm area, manure inputs can be excessively large increasing the risk of loss.

2.2.2.5 Weather and drainage

Nitrate loss is facilitated by the downward movement of water through the soil profile. As a result weather is the dominant factor determining the extent to which available N is lost (Goulding et al., 2000; Webb et al., 2000). During wet winters / in wetter parts of the UK concentrations tend to be lower but the quantity of N leached higher. While higher drainage increases leaching, larger quantities of water dilutes losses. As a result Johnson et al., (1997) observed higher concentrations during drier winters. The timing of rainfall also affects losses. Where drainage begins later in the autumn / winter crops are more established and have taken up more of the available N prior to the onset of leaching (Johnson and Smith, 1996). Highest concentrations are typically observed during early season drainage where residual mineral and fertiliser N is leached from post harvest soil (Goulding et al., 2000). However coincidence of rainfall and nutrient applications also results in high concentrations. Goulding et al., (2000) observed high concentrations where heavy late spring rain initiated drainage after spring fertiliser applications.

2.2.2.6 Soil type

The magnitude of N loss is also affected by soil type. Light sandy or shallow soils have low water retention capacities. As such they need little water to leach all nitrate from the soil profile resulting in high concentrations (e.g. Beaudoin et al., 2005). Predominately loamy or clay soils drain less easily, requiring more water to purge them of nitrate (Lord et al., 2007). The larger quantities of water associated with the leaching of nitrate from heavy soils dilutes leachate thereby reducing concentrations. As such sands and chalks have been identified as ‘leaky soils’ (Lord and Anthony, 2000). However in some instances the affect of soil type is more complicated. Macropores are more likely to develop in clay soils due to cracking, and over land flow (and associated N loss) more likely to occur on heavier soils where rainfall intensity exceeds lower infiltration capacities (Lord et al., 2007). Chemical soil properties are also important with high total N contents increasing the soils capacity to supply nitrate through mineralisation. Total N is lowest in sandy soils and highest in clay. However on grassland sites high total N is counterbalanced by lower mineralisation due to lack of soil disturbance and an established crop over winter (Lord et al., 2007).

2.2.2.7 Temperature

Mineralisation is more active where soil temperatures are higher (Smith et al., 2002). Temperature also affects crop growth; crop growth is rapid under mild conditions, but excessively high temperatures can induce drought conditions, reducing yields and increasing residual fertiliser available for loss as observed by Sieling et al., (1997).

2.3 Effects of nitrate on human health and ecosystems

2.3.1 Human health impacts

Nitrate in drinking water was widely believed to be responsible for methaemoglobinaemia in babies, and stomach cancer (Croll and Hayes, 1988). Nitrate is thought to be converted to nitrite which converts haemoglobin in the blood to an oxidised form. Oxygen is unable to bind to oxidised haemoglobin reducing the oxygen carrying capacity of the blood. As a result babies, which are less able to re-transform the oxidised haemoglobin to its original form and more vulnerable to its initial oxidation, become oxygen starved. A lack of oxygen gives them a bluish tinge hence the colloquial name 'blue baby syndrome' (Addiscott and Benjamin, 2004). With respect to stomach cancer, nitrite from nitrate reacts in the stomach with secondary amines formed in the digestion of meat or other proteins to produce a carcinogenic N-nitroso compound (Tannenbaum, 1987). As a result of health concerns the World Health Organisation (WHO) outlined nitrate limits in 1970. Concentrations less than 11.3mg N l^{-1} were deemed satisfactory, 11.3-22.6 mg N l^{-1} acceptable and greater than 22.6 mg N l^{-1} not recommended. Based on these standards the EU adopted a maximum admissible concentration for drinking water of 11.3mg N l^{-1} in 1980.

However in recent years the studies exposing these links has been re-evaluated and evidence suggests nitrates are less of a health concern than previously thought (Addiscott and Benjamin, 2004). The oxidation of haemoglobin is now thought to be caused by nitric oxide and not nitrate. Whilst this can also originate from drinking water, there is evidence that where methaemoglobinaemia has occurred, admissions to hospital were almost exclusively due to gastro-enteritis (Hegesh and Shiloah, 1982). The study found no correlation between methaemoglobinaemia and the ingestion of nitrate by infants. Large nitrate concentration observed in the blood instead reflect a defensive response to gastro-enteritis. Nitric oxide is produced by the body which is subsequently converted to nitrate on contact with oxidised haemoglobin. Gastro-enteritis is likely to have been contracted from polluted wells. These findings support toxicological tests performed on infants in 1948 which suggested nitrate alone, without bacterial pollution, does not cause methaemoglobinaemia (Cornblath and Hartmann, 1948). Furthermore the majority of methaemoglobinaemia cases corresponded with very high nitrate concentration, some in excess of 270mg N l^{-1}. Re-evaluations by L'hirondel and L'hirondel (2002) concluded that there is little evidence that nitrate is the prime cause of methaemoglobinaemia, while Avery (1999) considered bacterial pollution not nitrate to be probably responsible for the condition.

The links between nitrate and stomach cancer have also failed to withstand the scrutiny of re-evaluation. Studies by Forman et al., (1985), Beresford, (1985) and Van Loon et al., (1998) found no relationship / a negative relationship between nitrate concentrations and stomach cancer. As a result the UK government formally accepted the absence of a link in 1985 after Acheson (1985).

Contrary to previous nitrate concerns, there is now evidence to suggest that nitrate offers health benefits. Nitrate is produced and retained by the body and converted to nitrite, an important anti microbial agent. In the presence of stomach acid and nitrite, common causes of stomach and bowel problems including gastro-enteritis can be killed. However it seems unlikely that drinking water standards will be altered to reflect current understanding of nitrate and human health.

2.3.2 The effect of nitrate on aquatic ecosystems

Nitrate occurs naturally within in aquatic systems but where concentrations rise above baseline conditions, freshwater and marine ecosystems are adversely affected (Heathwaite, 1993). Nitrogen, alongside phosphorus (P) acts as a rate limiting nutrient in aquatic ecosystems (Heathwaite, 1993). But while its presence is essential for primary productivity, high concentrations induce eutrophication.

Eutrophication is a process not a state, defined in the Urban Waste Water Treatment Directive (91/271/EEC) and OSPAR agreements as 'The enrichment of water by nutrients, especially compounds of N and / or P, causing an accelerated growth of algae and higher forms of plant life to produce undesirable disturbances to the balance of organisms present in the water and to the quality of the water concerned'. More details of the symptoms associated with eutrophication are shown in Figure 2-5.

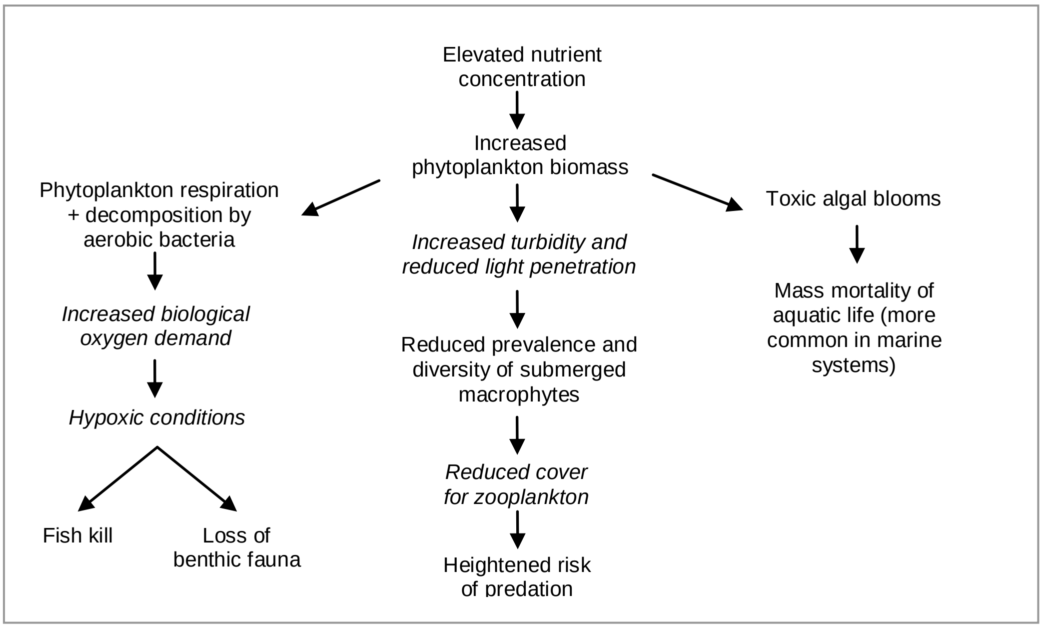


Figure 2-5: The effect of eutrophication on aquatic ecosystems

Differences in nutrient limitation and physical characteristics mean responses to elevated nitrate concentrations differ between rivers, lakes and estuarine / coastal environments. Freshwater systems are typically phosphorus limited meaning phosphorus enrichment is more likely to induce eutrophic conditions than nitrate; however nitrate enrichment is thought to exacerbate the problem (Newmann et al., 2005). Longer retention times means lakes are more susceptible to eutrophication than flowing waters, with problematic algal growth associated with nitrate concentrations as low as 1.5mg N l^{-1}. Significant accumulations of algal blooms are less likely in rivers where algae is flushed out faster than it can grow (OECD, 1982) and organisms exposed to high nutrient concentrations for shorter periods of time. However recent studies suggest flowing waters are more sensitive to excessive nutrient loadings than previously thought. Nutrient limitation of algal growth in flowing waters is now thought to be widespread, and thus of international concern (Smith et al., 1999).

Nitrogen plays a more critical role in the eutrophication of coastal and estuarine systems than freshwater systems (Kelly, 2001). However, with rivers serving as rapid conduits for excessive nutrient loadings to estuarine and coastal systems, inputs to both freshwater and marine environments should be minimised. In contrast to the long standing concern and attention surrounding the eutrophication of freshwater bodies, the nutrient status of coastal water and estuaries is a more recent concern (Owens, 1993). Understanding the links between nutrient enrichment and coastal and estuarine waterbody state is complex. Marine systems are confounded by longer term changes operating at ocean basin scale e.g. meteorological and circulation factors whilst estuaries reflect a complex interaction of freshwater and marine systems (Owens, 1993). In spite of this there is considerably evidence to suggest high nitrate concentrations are driving eutrophication in coastal and estuarine environment (e.g. Cadee 1990; Radach et al., 1990) and nuisance algal growth has been observed in coastal zones and seas throughout Europe (OSPARCOM 1992; Kronvang et al., 1993).

Despite higher nitrate concentrations in rivers than lakes, lakes have been the focus of eutrophication research. Only recently have concerns shifted towards flowing water and indeed estuarine / coastal waters. The dynamic nature of flowing water makes it difficult to isolate nutrient impacts from that of other, mostly physical, factors (Newmann et al., 2005). Despite being the basis of legislation, the relationship between water chemistry and ecological quality is vague (Donohue et al. 2006). Due to the complexity of the interactions between nutrients, water bodies and eutrophication, few empirical thresholds exist (Haygarth et al. 2005). The WFD demands that water bodies reach good ecological status, necessitating an increased and quantified understanding of the links between catchment attributes (morphology, geology, land use), water chemistry and the ecological status of aquatic environments (Donohue et al. 2006). While there are many approaches and indices for the assessment of biological / ecological quality, there is a need for greater standardisation (Hering et al., 2010). Implementation of the WFD has resulted in co-ordinated action to produce a harmonised classification system for ecological and biological state (UKTAG, 2010). Assessments will be more comparable between studies and countries and allow changes / improvements in ecological state to be tracked.

2.4 Legislation and N loss

2.4.1 Evolution of European environmental policy

European environmental legislation was first introduced in 1973 and aimed to protect human health and ensure international trade was not affected by inconsistencies in environmental policy (Kallis and Butler, 2001). Legislation was based on water quality standards and the regulation of permissible levels of pollutant discharge. However by the late 1980’s increasing environmental concern regarding agricultural pollution, urban waste water and ecological condition resulted in second wave of European water legislation which included the Urban Waste Water Directive (91/271/EEC) and the Nitrates Directive (91/676/EEC). However in 1995 a ‘state of the environment’ report highlighted continued deterioration of water quality, particularly from diffuse pollution (Kallis and Butler, 2001). Existing legislation was considered too fragmented, necessitating a fundamental re-think of European water policy (EC, 2010b). In response the European Commission proposed a more stringent, integrated, streamlined policy framework offering protection to all waterbodies through a combined approach of emission limits and quality standards, with greater public participation (EC, 2010b). This is now known as the Water Framework Directive.

2.4.2 The Water Framework Directive

The Water Framework Directive, implemented in 2000 represents the most comprehensive European environmental legislation / regulation to date. It aims to protect and enhance the aquatic environment, prevent further deterioration, and promote sustainable water management to ensure the long term protection of water resources (Defra, 2001a). The WFD was introduced to address the inadequacies of previous legislation which offered insufficient protection to aquatic ecosystems, particularly from diffuse pollution sources. Unlike previous legislation the WFD provides a co-ordinated approach to the protection of aquatic environments, integrating extensive existing legislation. The WFD goes beyond pollution control, instead ensuring the integrity of ecosystems as a whole and recognising interactions between land and water management. Rather than being implemented on the basis of administrative or political boundaries, the WFD respects the natural geographical and hydrological unit of river basins (EC, 2010b).

The Water Framework directive requires all waterbodies to achieve 'good' and 'non-deteriorating' status by 2015. For surface waters good status refers to both ecological and chemical status, whilst for groundwater, good chemical and quantitative status is required. Good ecological status describes water bodies showing only slight departure from conditions that would be expected under minimal anthropogenic impact, as reflected in biological, hydro-morphological and physio-chemical conditions (Figure 2-6). Good chemical status requires adherence to existing EU water quality and pollution control standards, and compliance with additional restrictions set out in accompanying annexes, and a new daughter directive addressing priority substances (2008/105/EC). In addition groundwater bodies must not damage the surface water bodies or terrestrial ecosystems into which they exfiltrate. To reach good quantitative standard, long term average abstraction must not exceed the available resource (2000/60/EC). Further clarification of groundwater objectives are to be contained in a groundwater daughter directive.

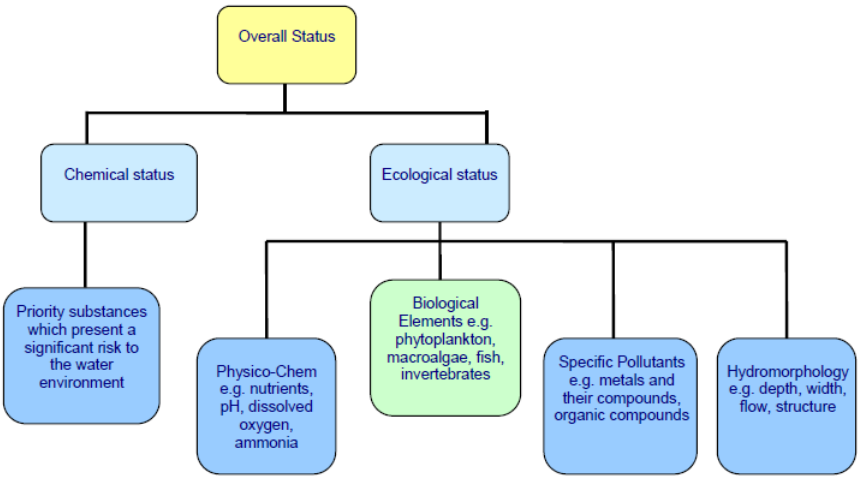


Figure 2-6: Components of 'overall status' for surface water bodies (Environment Agency, 2009)

Artificial or heavily modified waterbodies (those created by humans or subject to physical alteration by human activity which substantially changes its hydro-geomorphological character) are subject to slightly different requirements. Where compliance with good status would induce significant adverse effects on the wider environment or on activities important for sustainable human development e.g. power generation, navigation and flood protection, waterbodies are required to reach good ecological potential. Artificial / heavily modified waterbodies are likened to the closest natural aquatic ecosystem and must achieve the standards appropriate for that waterbody. In doing so hydrogeomorphological characteristics of the waterbody are respected and adverse effects on the specific uses and the wider environment avoided (Borja and Elliot, 2007).

While compliance with chemical standards is relatively straight forward, defining good ecological and in particular biological status has proved more challenging. Implementation of the WFD has required harmonisation of good ecological status between member states, however due to differences in environmental conditions and species, some felt this was comparing the incomparable (Hering et al., 2010). An inter-calibration process has been undertaken in an effort to harmonise results but not assessment systems (EC, 2010c). UKTAG has developed methodologies for the assessment of biological quality elements in UK water bodies (UKTAG, 2010).

Unlike previous legislation the WFD contains no standards which define these goals instead referring to standards in existing legislation. For example nitrate concentrations must be less than 11.3mg N l^{-1} in line with the maximum admissible concentration detailed in the Drinking Water Directive (98/83/EC). Similarly nitrate vulnerable zones, implemented as part of the Nitrates Directive (91/67/EEC) must be respected. While the WFD provides common goals it does not identify measures which must be adopted to achieve them. Member states are instead able to implement measures in the most adequate and efficient way with respect to local environmental and socio-economic conditions (Kallis and Butler, 2001). However despite regional differences in its implementation, the WFD encourage co-ordination and co-operation between member states to address shared technical challenges. As a result member states and the European Commission have agreed on a Common Implementation Strategy (CIS) to support the clarification and development of technical and scientific information necessary for the practical implementation of the WFD (EC, 2001).

The WFD is organised at the river basin scale highlighting an appreciation of the interactions between land management and water body status. Compliance with the WFD requires submission of six yearly River Basin Management Plans (RBMPs) documenting the implementation process and the achievement of environmental objectives. The first of these was due in 2009. Implementation of the WFD was initially concerned with the characterisation of river basins (physical and chemical conditions) and assessment of pressures and impacts (Defra, 2001b). In doing so waterbodies at risk of failing WFD objectives could be identified and reference conditions devised. In the UK this initial stage of characterisation was conducted by the Environment Agency in 2003-2004 and is being followed by a second, more targeted assessment. The latter will make use of more recent monitoring data and improved understanding of ecological indicators connected with ecological status (Defra, 2001b). Where waterbodies are at risk of not meeting good status programmes of measures have been devised and should be implemented by 2012. Measures will address the specific pressures jeopardising achievement of environmental objectives in each river basin. For example in the UK's South West River Basin measures will target agriculture, mining, point sources from sewage works and trade industry, abstraction and aspects of physical waterbody modification (Environment Agency, 2009). Monitoring requirements set out by the WFD require the impact of PofMs to be assessed, reported and refined as necessary. Details of revised programmes of measures will be included in the next RBMPs due to be published in 2015. The iterative processes of assessment and refinement will ensure intervention is cost effective. Economic analysis and full public participation under pins the River Basin Management planning process.

2.4.3 Other legislation

2.4.3.1 Nitrates Directive

The Nitrates Directive (91/676/EEC) aims to reduce nitrate loss from agricultural sources and protect ground and surface water quality by promoting good agricultural practice (EC, 2010d). It was development in response to growing environmental concerns surrounding the impact of nutrient enrichment; prior to this legislation existed to primarily protect human health (Kallis and Butler, 2001). It is now an integral part of the WFD and represents one of the key instruments in the protection of waters against agricultural pressures. Compliance with the Nitrates Directive is a pre-requisite to compliance with the Water Framework Directive.

Implementation of the Nitrates Directive required identification of waterbodies (surface and groundwater) containing, or at risk of containing, nitrate in excess of 11.3mg N l^{-1}, or those considered, or at risk, of becoming eutrophic. 11.3mg N l^{-1} represents the maximum admissible concentration as permitted by the Drinking

Water Directive (98/83/EC); alike to the WFD the Nitrates Directive does not contain its own mandatory standards, instead referring to relevant limits in existing legislation. The land draining these waterbodies is designated as Nitrate Vulnerable Zones (NVZ) within which Action Programmes are implemented by farmers on a compulsory basis. Farmers were required to follow Codes of Good Agricultural Practice (which are also implemented on a voluntary basis on none NVZ sites), adhere to closed periods of manure spreading, have sufficient manure storage to comply with closed periods, balance fertiliser applications with crop demand and soil supply, and limit manure applications to 170kg N ha^{-1} (Goodchild, 1998; Defra 2002a; EC, 2010d).

The UK originally designated 66 NVZs covering 600000ha (8% of England) in 1996, with action programmes implemented from 1998 (Figure 2-7). However this was increased by 47% to 55% of the country in 2002 to extend protection to all surface and groundwaters and not just drinking water (Defra, 2002a). Assessments of the effectiveness of Action Programmes have since highlighted that further action is required to reduce nitrate pollution, resulting in a further extension which increased the NVZ area in England to 62% in 2009 (Lord et al., 2007; Defra, 2010a). NVZs now cover 39.3% of Europe, and of the original EU-15, NVZs have increased by 1% between 2000-2003 and 2004-2007 (EC, 2010a).

2.4.3.2 Urban Waste Water Treatment Directive

The Urban Waste Water Treatment Directive (UWWTD) (91/271/EEC), also introduced in 1991, aimed to tackle pollution from urban waste water, namely discharge from sewage treatment works, which are high in nutrients. The directive specified maximum nitrate concentrations in discharge from sewage treatment works according to population size and the sensitivity of receiving waters (EC, 2010e). Where sewage works treat effluent from populations between 10,000 and 100,000 persons nitrate concentrations must not exceed 3.4mg N l^{-1}, whilst those in agglomerations greater than 100,000 are limited to 2.3mg N l^{-1} (Skeffington, 2002). The UWWTD has therefore been effective in regulating point sources, reducing nitrate concentrations in larger rivers downstream of urban areas. However it does little to address diffuse pollution.

a)

b)

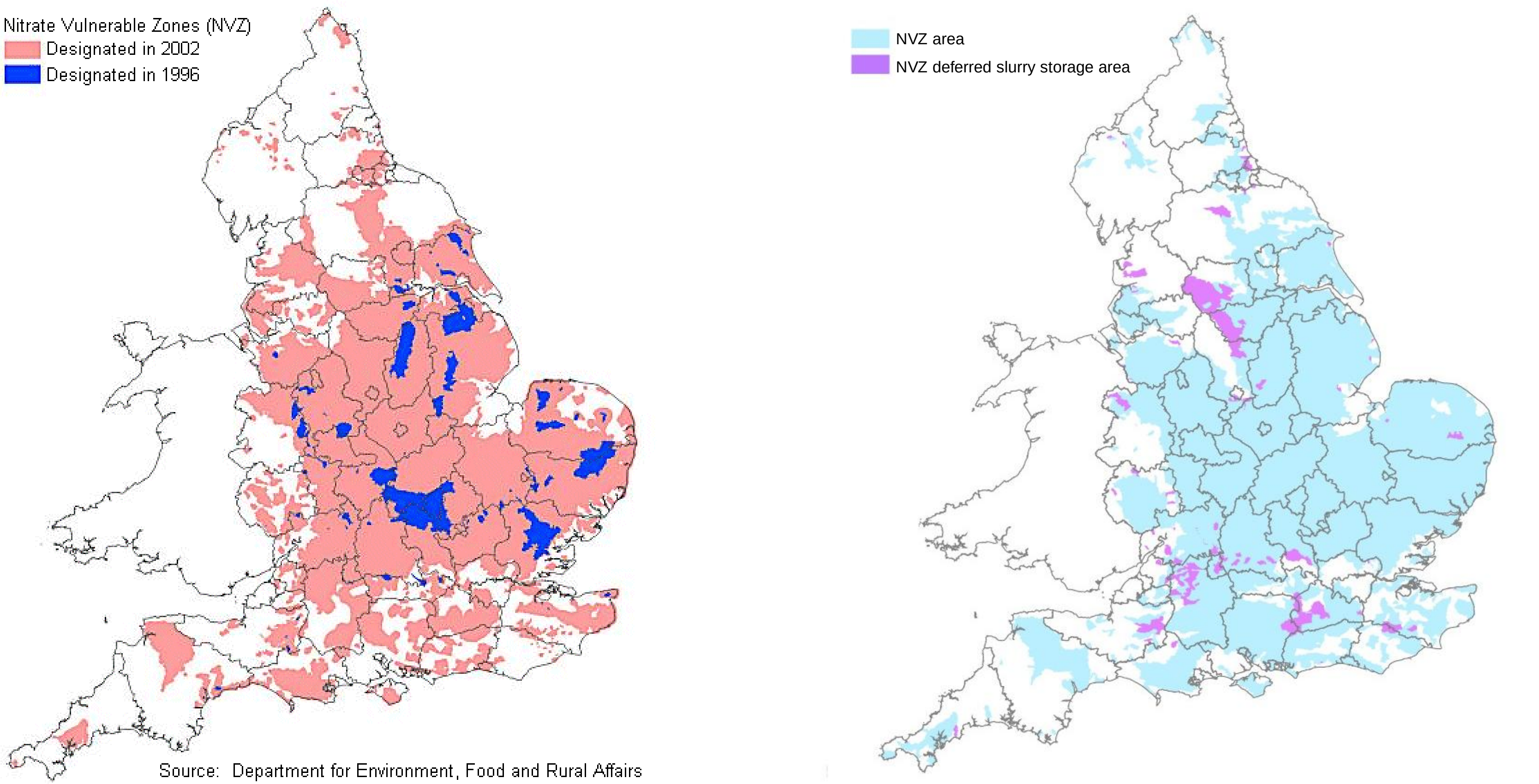


Figure 2-7: The development of Nitrate Vulnerable Zones (NVZ) in England and Wales. NVZs in a)1996 and 2002 (Environment Agency, 2010) and b) 2010 (after appeals regarding extensions in 2009) (Defra, 2010a).

2.4.3.3 Other relevant legislation and European policy

The Habitats Directive (92/43/EEC), Bathing Waters Directive (previously 1976/160/EEC but recently updated to 2006/7/EC), Fish Breeding Directive (2006/44/EC) and the Integrated Pollution Prevention and Control Directive (IPPC) (96/61/EC replaced by 2008/1/EC) also affect water quality and are of indirect benefit to nitrate pollution. Similar to the UWWTD the IPPC has been effective in addressing urban point sources. The Habitats and Bathing Waters Directives have also attempted to control nutrient concentrations, however bathing water standards continue to be breached as a result of diffuse pollution. The Fish Breeding Directive has however been a key driver in nutrient controls. In addition, reforms of the Common Agricultural Policy are likely to impact on nitrate use and thus on nitrate loss from agricultural land. Subsidies are no longer paid on a production basis, instead offering 'Single Farm Payments' for land under agricultural management where environmental (amongst other) standards are met. To receive payment all farmland must be kept in 'good agricultural and environmental condition', known as Cross Compliance, targeting soils, habitats and landscape features (Defra, 2010b). Protection of soils and associated improvements in soil structure may improve nutrient utilisation. In addition decoupling subsidies and production removes the incentive to farm as intensively. Nitrate loss is typically lower where farming is less intensive.

2.4.3.4 Non regulatory drivers

In response to concerns surrounding diffuse agricultural pollution and the need to achieve challenging legislative targets, a range of voluntary initiatives and schemes have been introduced aimed at reducing diffuse pollution and increasing environmental awareness. Most noteworthy are the Catchment Sensitive Farming Delivery Initiative and Environmental Stewardship schemes.

The England Catchment Sensitive Farming Delivery Initiative (ECSFDI) is a Defra funded initiative to tackle diffuse water pollution from agriculture, implemented in partnership with Natural England and the Environment Agency. The programme was introduced in 40 priority catchments throughout England in 2006. In 2008 a further 10 catchments were added and extensions made to 7 of the existing catchments (Defra, 2007). Catchment Sensitive Farming (CSF) aims to deliver practical solutions to farmers, promoting good land, nutrient, and livestock management as a means to reduce diffuse agricultural pollution. Emphasis is on the provision of advice through dedicated CSF officers who working closely with farmers to improve land management. In additional capital grants are made available for small scale infrastructural changes beneficial to diffuse pollution.

Environmental stewardship schemes provide funding to farmers to deliver effective environmental management, building on the earlier success of the Environmentally Sensitive Areas and Countryside Stewardship schemes (Natural England, 2010). Farmers may enter into different levels of the scheme according to the complexity / severity of measures adopted. Entry Level Stewardship ensures a basic level of land stewardship, but goes beyond that required for ’Cross Compliance’. The Higher Level Scheme offers more complex management options tailored to local conditions; this is best suited to priority areas and situations where maximum environmental benefit can be achieved thereby offering good value for money. A wide range of management options are included in the schemes however those addressing fertiliser, manure and soil management are of particular relevance to nitrate loss.

2.5 Mitigation methods

Diffuse agricultural pollution must be reduced in order to comply with legislative requirements and to minimise the adverse impacts of agriculture on ecosystems. While legislation and policy identifies the means through which this might be achieved, specific actions and intervention are required to reduce the losses from agricultural land. A wide range of ‘mitigation methods’ have therefore been developed that can be adopted on farm to reduce diffuse agricultural pollution. Cuttle et al., (2006) catalogues a collection of 44 methods however this list is not exhaustive. Mitigation methods target soil, livestock, manure and fertiliser management, and farm infrastructure, recognising the interactions between land management and water quality. Their implementation reflects an integrated approach to catchment management which is being adopted in England and Wales. The WFD is driving a similar approach throughout Europe.

Nitrate loss depends on the availability of N and the presence of transport processes to facilitate leaching. Since intrinsic risks associated with climate and soil type cannot be managed, mitigation targets the availability of N prior to the onset of drainage i.e. it addresses the problem at source. This is in contrast to phosphorus and sediment loss mitigation which tends to address the mobilisation and delivery of pollutants to waterbodies. Mitigation methods tackle aspects of nutrient, land and livestock management which increase the risk of nitrate loss (see section 2.2.2). Table 2-1 details the mechanisms and target areas associated with a selection of nitrate mitigation options focusing on those implemented as part of the WAgriCo project (see section 3.5.1 for more details). However a comprehensive 'Users Manual' of mitigation methods provides rationales and underlying mechanisms for the 44 aforementioned mitigation methods (Cuttle et al., 2006).

As an integral component of Programmes of Measures and the Nitrate Directives Action Programme, mitigation methods must be effective and efficient in reducing inputs and / or emissions from agriculture if waterbodies are to achieve good status. Cuttle et al., (2004) provides a detailed review of the effectiveness of a wide range of mitigation methods. However in keeping with the WFD's emphasis on cost effective action, the price of achieving such reductions must also be considered. The Cost Curve project provided estimates of the cost and effectiveness of a wide range of mitigation methods at the farm scale (Haygarth et al., 2005; Cuttle et al., 2006). Low cost methods were found to be most cost effective, however as many represented good agricultural practice which is already adhered to by most farmers, their impact is likely to be reduced. This highlights the importance of baseline conditions in determining the effectiveness of mitigation. The acceptability of mitigation is also important, especially where voluntary initiatives are adopted - uptake can itself been considered a measure of mitigation effectiveness (Lord et al., 2007). In a Swedish study farmers were found, perhaps not surprisingly, to favour measures that looked good over those that affected their farming practices, and did not want to have to spend nor lose money (Ulen and Kalisky, 2005). Farm type, structure and crop rotations will limit the applicability and thus uptake of some methods. However where mitigation is compulsory, ease of implementation and enforcement should also be considered. Given the challenging timescales over which good status is to be achieved, timescales of implementation are important. In addition potential for pollution swopping (reducing emissions of one pollutant at the expense of another) and the level of certainty must be considered when comparing mitigation options.

Table 2-1: Mitigation methods to reduce N loss from agriculture - Mechanisms, cost, timescales, farmer acceptance and effectiveness of a selection of mitigation options based on results from the DEFRA commissioned Cost Curve Project (Haygarth et al., 2005). A = arable, G = grassland systems; some mitigation methods were not applicable to both arable and grassland systems. [a] denotes mitigation implemented as part of the WAgriCo project – see section 3.5.1 for details.

Target	Mitigation method	Mechanism		Cost 0-4 (4 highest)	Timescales 1-4 (4 max)	Acceptability 0-2 (2=acceptable)	Effectiveness 0-4 (4 max)
Fertiliser	Adopt of fertiliser recommendation system [a]	Fertiliser recommendations ensure fertiliser inputs do not exceed crop needs once soil N supply, manure inputs, soil type and climate have been considered. Leached losses are high where fertiliser is applied above the economic optimum.	A/ G	0	1	2	2
	Integrate fertiliser and manure N [a]	Accounting for crop available N in manures avoids excessive applications of fertiliser N. Many farmers do not account fully for N applied as manure.	A:	0	1	1	2
			G:	0	1	1-2	1-4
	Reduced fertiliser application rates [a]	Limiting fertiliser N reduce the quantity of residual nitrate in the soil after harvest. Reductions are most effective were fertiliser was previously applied at supra optimal levels. However reduced fertiliser has no effect on the amount of nitrate mineralised from soil organic matter which represents a larger pool of N available for leaching over the autumn and winter.	A:	1	1	0	2
			G:	2	1	1	4
	Avoid spreading fertiliser at high risk times [a]	Fertiliser applications should be avoided where crop uptake is low, the risk of leaching high, and / or the potential for rapid transfer of N from the soil surface high (e.g. when soils are saturated, frozen or snow covered). In doing so N is more utilised and / or less at risk of loss.	A:	0	1	2	2
			G:	0-2	2	1	4
Land use	Convert arable land to extensive grassland	Reduced fertiliser inputs and continuous vegetation cover reduces the risk of loss. Immobilisation into soil organic matter provides a sink for available N. Grassland avoids frequent cultivation which stimulates mineralisation.	A:	3	1	0	4
Livestock	Reduce stocking rates	Reduced stocking reduce the amount of N deposited in fields as excreta and handled in manures. This will ease pressure on manure storage and provide greater flexibility in avoiding manure applications at high risk times. Reduce stocking also reduces the frequency of high nitrate urine spots, and reduce fertiliser inputs.	A:	3 / 4	2	0	2
			G:	3	1	0	1-3
	Reduce dietary N intake	Reducing N intake will reduce N in excreta / handled in manures (see above for benefits). In many cases recommended intakes are exceeded meaning reductions would have little impact on growth or milk production.	A:	2	2	1	1
			G:	1	1	1	1-2
Manure	Avoid spreading manures / slurry at high risk times [a]	Manure and slurries should not be applied when there is a risk of surface run off e.g. where soils are saturated, frozen or heavy rain is expected. Surface run off soon after manure applications rapidly transports N to waterbodies. Slurries and poultry manures should not be applied where crop demand is low, in autumn / early winter when the risk of leaching is high and temperatures high enough for mineralisation, and where there is a risk of N being rapidly transported to field drains e.g. where soils are dry and cracked above drains.	A:	0	1	1	3/2
			G:	0-3	1	1	1-2
Soil	Establishment of cover crops [a]	Cover crops increase uptake of residual N post harvest and mineralisation inputs. Less N is subsequently available for leaching over winter.	A:	1	1	1	2

Of the mitigation methods presented in Table 2-1 arable reversion, correct timing of manure applications, reduced fertiliser and the integration of manure and fertiliser supply represent the most effective means of tackling nitrate loss. Of these only arable reversion is considered prohibitably expensive, with accounting of manure N likely to reduce fertiliser costs. However a reduction in fertiliser is generally unappealing to farmers. Similarly while reductions in stocking rates are effective, the measure is considered unacceptable by farmers. For some mitigation methods acceptance requires greater understanding. For example livestock are often fed more N than they can effectively utilise. A reduction in dietary N would reduce the risk of N loss at no detriment to farm profitability. However the method is current considered unfavourable to farmers. Ultimately which methods are employed is likely to depend on the level of regulation attached to their implementation and the availability of compensation.

2.6 Assessing mitigation effectiveness

2.6.1 Why assess mitigation effectiveness?

Commitment to water quality and ecological standards under the WFD has generated a need to assess the impact of mitigation and quantify effectiveness. The WFD requires responses to PofMs to be monitored and progression towards targets be documented. In doing so assessments provide assurance that mitigation has been appropriately targeted to local conditions, and that environmental objectives will be attained within available timescales. Assessment is complementary to the iterative nature of the WFD; where PofMs are ineffective or insufficient mitigation can be revised, and where they are found to be effective, assessments justify wider adoption. Re-evaluation ensures mitigation is increasingly targeted and thus cost effective. The WFD represents a key driver in improving the availability of assessment and monitoring methods (Collins and McGonigle, 2008).

2.6.2 Requirements of an assessment method

Assessment methods must provide evidence of a reduction in nutrient loadings / concentration or an ecological response over an appropriate timescale at a scale of relevance. The WFD requires RBMPs on a 6 yearly basis and good status by 2015; short to medium term assessments are therefore important. With environmental objectives targeting waterbodies, and PofMs devised and implemented at the river basin scale, quantification of catchment responses are of particular relevance. However under some circumstances field and farm scale assessment may be favoured where progress can be tracked in the short term.

Assessment methods must recognise that responses to mitigation are often site specific and depend on the interrelationship between nutrient pressures and inherent environmental vulnerability of the landscape. In accordance with Zalidis et al., (2004), assessment methods must integrate physical, chemical and biological processes. They must also be sensitive to the wide range of mitigation methods available and respect the scale at which they are implemented. Sensitivity to mitigation methods which are policy relevant (e.g. those included in NVZ Action Programmes) is of particular importance.

In keeping with the WFD's drive for cost effectiveness, choice of assessment must respect the availability of resources and data availability. Ideally assessment will complement wider WFD monitoring requirements. Assessment methods must be practical and suited to end users. Communicable, understandable approaches will ensure implementation of the WFD is interactive and permit public / stakeholder involvement. Methods increasing awareness of nutrient loss and encouraging further intervention would be particularly useful. However from a control and enforcement point of view, it is important that responses can be attributed to farm management. Indeed Van der Werf and Petit (2002) consider sensitivity to farm practice more important than sensitivity to environmental factors when assessing agri-environmental performance.

2.6.3 Previous assessments of mitigation effectiveness

To support the implementation of mitigation in Nitrate Directive Action Programmes and other agri-environmental schemes, numerous evaluations of mitigation effectiveness have been performed. However investigations have largely been confined to the field and farm scale (e.g. Johnson and Smith, 1996; Shepherd, 1999; Johnson et al., 1997 and 2002) and performed using designed field experiments (e.g. Beckwith et al., 1998) or derived from simulations of modelled farms (e.g. Haygarth et al., 2005; Cuttle et al., 2006). Despite being more relevant to the waterbody focus of the WFD, fewer evaluations have been conducted at the catchment scale because of the difficulty in covering a wide range of environmental and agricultural conditions. As concluded by Haygarth et al., (2005) there remains a need to quantify the effectiveness of mitigation adopted in practical field and catchment context, especially in integrated, complex landscapes. Suitable assessment methods are required if gaps in the literature are to be addressed. Shortcomings of traditional monitoring approaches provide an opportunity to explore new monitoring tools (Collins and McGonigle, 2008). Existing and new assessment methods are discussed presently.

2.7 Assessment methods

A range of different assessment method exists including measurement, nutrient accounting approaches and modelling, however their popularity and applicability varies. The concepts under pinning each approach and their usefulness as evaluators of mitigation effectiveness are discussed below. Investigation of model based assessment methods were beyond the scope of this research programme, however it was deemed appropriate to include some background information to place measurement and budget based discussions in the wider context of available / potential assessment methods.

2.7.1 Measurement

Long time series of N concentrations and loads in water bodies or biological surveys potentially provide the best analysis of mitigation success describing the actual change in chemical quality or ecological functioning following the implementation of mitigation. Responses are quantified by measurements, uniquely reflecting all influential environmental processes and conditions affecting the method’s effectiveness. However, it is not always clear why any positive or negative response to mitigation has been achieved because of the complex array of environmental processes involved and our incomplete knowledge of nutrient dynamics, especially in-stream processes and ecological responses. Variations in weather (and the influence on losses) between years adds another layer of complexity and makes it difficult to distinguish the effect of the mitigation method from environmental noise (Lord et al., 2002; Bechmann et al., 2005; Lord et al., 2007). In the medium-term, the continued development and installation of automated *in situ* sampling and analytical equipment facilitating high frequency sampling will help improve our understanding and provide more representative assessments (Harris and Heathwaite, 2005).

In catchments, datasets must extend over many years, have a high spatial and temporal resolution, and include representative sample sites if they are to reveal responses to changes in catchment management and why they have occurred. Collecting these data at the resolution required is expensive and often logistically difficult. Unsurprisingly, detailed monitoring at the necessary resolution typically takes place in only a selected number of locations and few countries have the comprehensive long-term records required for mitigation evaluation (Vagstad et al., 2004). While the River Basin Management Plans demanded by the WFD may increase data availability, emphasis on cost effectiveness and the avoidance of disproportionate costs by the WFD may be a strong argument for the development of lower cost alternatives for mitigation evaluation.

An important consideration regarding the use of measurement as an assessment method is the timescale over which records need to be collected. Time lags observed between changes in agricultural practice / implementation of mitigation methods and reductions in nutrient concentrations frequently exceed the timeframe for achieving good ecological status under the WFD (Stalnacke et al., 2003; Vagstad et al., 2004; Granlund et al., 2005; Grizzetti et al., 2005; Kronvang et al., 2005). Silgram et al. (2005) estimated it would be 60 years before 50% of the impact of Nitrate Sensitive Area (NSA) mitigation methods would be measurable in England. A recent review of the Nitrate Vulnerable Zone Action Programme calculated a 58-131 year delay before borehole nitrate concentrations decrease to 50% of their initial value (Hughes et al., 2006). Essentially these delays are a result of hydrological time lags and catchment buffering.

Lengthy transit and residence times associated with permeable groundwater dominated catchments delay the arrival of low nutrient flows into water bodies. The slow progression of low nutrient water through groundwater catchments was demonstrated by Silgram et al. (2005) who continued to observe high nitrate concentrations in groundwater despite a reduction in soil root zone leachate concentrations following the implementation of mitigation. Groundwater systems comprise multiple flow pathways (van Lanen and Dijksma, 1999), each with a unique transit time that reflects aquifer porosity, permeability, hydraulic gradients and geometry (Wriedt and Rode, 2006). Older nitrate rich water may mix with younger, lower nitrate waters to create water of an intermediate concentration. Nitrate may also diffuse between mobile and immobile water, retarding its movement through the bedrock (Molenat and Gascuel-Odoux, 2002). Impermeable catchments respond faster (Vagstad et al., 2004) with changes in farm practice and the resulting reductions in overland flow concentration rapidly translated into a reduction in the nutrient enrichment of water bodies. Over both permeable and impermeable catchments, responses are slower where land is flatter (Grizzetti et al., 2005) and where nutrients are applied further from water bodies. Variation in catchment response rates are therefore a reflection of hydrological and hydrogeological differences (Iital et al., 2005).

Soil, sediment and biological retention and re-release can dampen responses to changes in agricultural practice and even counteract mitigation efforts. Buried organic matter in stream sediments sequesters nitrates until mineralised (Grizzetti et al., 2005), and where anaerobic conditions prevail (saturated soils and groundwater) denitrification may buffer nitrate concentrations with the potential to degrade up to 90% of the initial concentration (Wendland et al., 2005; Wriedt and Rode, 2006). Denitrification has a half-life of 1-5 years meaning attenuation is maximised where transit times are long (Kersebaum et al., 2003). The implications of catchment inertia and lake resilience must be considered when assessing the responses to mitigation using measurement (Stalnacke et al., 2003; Granlund et al., 2005; Kronvang et al., 2005).

Measurement presents a more viable assessment option at the field and farm scale where the edge of field / farm outflow export is measurable and the range in environmental processes or noise may be less significant (e.g. Beckwith et al., 1998; Shepherd, 1999). Farm scale evaluations also provide an insight into the applicability of mitigation to the whole farm system (e.g. Johnson et al., 2002). At the catchment scale, resource and money often limit sampling to catchment outlets only. While this allows multiple mitigation methods across the catchment to be evaluated, results do not explicitly reveal which loss processes are targeted or are most sensitive to a mitigation option, or whether combinations of methods are indeed complementary. It is not possible, however, to substitute more extensive catchment monitoring with upscaled field evaluations due to the spatial and temporal variability in nutrient sources, losses, and delivery pathways. Contributions from sources active only at catchment scale such as bank erosion and local point sources are absent from observations made at the field scale. Variability in weather and environmental characteristics also means replicas of farm and catchment studies are not possible.

Direct measurement represents a useful means of assessment, but one that is not possible in all situations. Where monitoring is not cost effective and delayed responses require long time series to reveal the impact of mitigation, alternatives are needed. Nutrient budget methodologies and models provide a range of alternative approaches.

2.7.2 Nutrient budgets

Nutrient budgets are most commonly used to quantify nutrient management by evaluating inputs and outputs over a defined time period. Using simple accounting procedures often automated by spreadsheets and user friendly interfaces, the nutrient surplus or deficit is calculated. System boundaries are flexible, resulting in a range of methodologies applicable from plot to national scale including farm gate, soil surface and soil system budgets.

Farm gate budgets quantify nutrients that enter and leave the farm gate with no consideration of internal transfers or loss processes (Figure 2-8a). Only nutrients that are bought and sold and are subsequently imported and exported on and off the farm are included. Soil surface budgets account for nutrients entering and leaving a field via the soil surface including denitrification and ammonia volatilisation (Figure 2-8b). Soil system budgets include all inputs and losses including immobilisation, runoff, leaching, denitrification and ammonia volatilisation, providing a detailed understanding of nutrient fate (Figure 2-8c). Surpluses represent those nutrients not accounted for by loss processes and are therefore balance specific. The purpose of the study will often define the type of budget used, although the decision is frequently constrained by data availability (Oenema et al., 2005). Farm gate balances, for example, demand minimal and routinely available data compared with soil system balances. Typically, input requirements are smaller than for models and can be obtained relatively easily from increasingly computerised farm records.

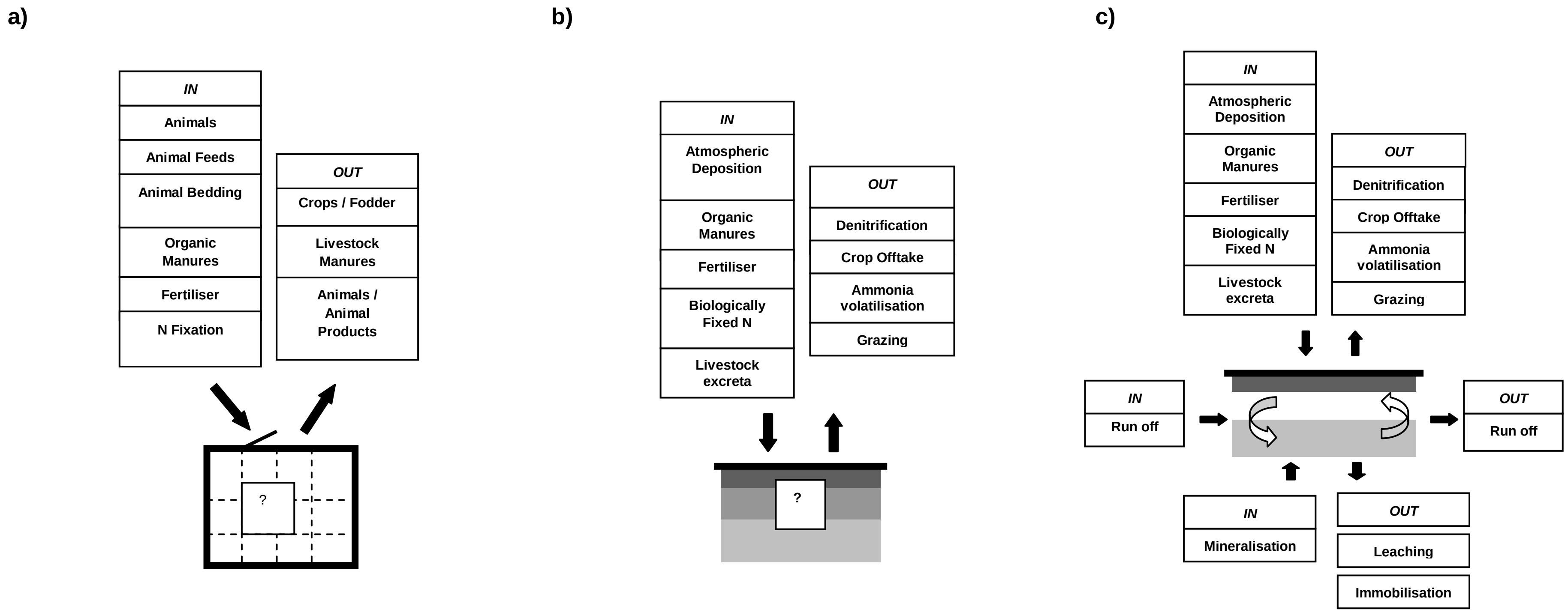


Figure 2-8: Nutrient budget methodologies. Inputs and outputs associated with a) farmgate b) soil surface and c) soil system nutrient budgets.

Nutrient budgets reveal changes in the distribution of nutrients between pools (Vagstad et al., 2004), encouraging an awareness of nutrient management and highlighting the benefit of internal nutrient cycling (Brouwer, 1998; Oenema et al., 2003). They provide a means of assessing nutrient efficiency, identifying areas where the potential for nutrient surpluses and loss is high, and thus where mitigation should be targeted. The financial and environmental consequences of inefficient nutrient use are exposed, communicating the need to adopt mitigation to minimise both (Brouwer, 1998; Lord et al., 2002) and promoting farmer accountability. Budgets are responsive and as such can be used to monitor changes in nutrient management, and quantify performance against other farms and target values. As a means to improved nutrient management, nutrient budgets are an accepted and commonly used tool.

Over 50 nutrient accounting systems are currently in use across EU member states (Goodlass et al., 2003). The Oslo and Paris Commission (OSPARCOM) utilises farm gate budgets to monitor N and P emissions into the North Sea (OSPARCOM, 1994), while the Organisation for Economic Co-operation and Development (OECD, 2002) recognises the gross soil surface balance as an effective agri-environmental indicator. In England and Wales, nutrient budgets have recently become a feature of nutrient management with Planning Land Application of Nutrients for Efficiency and the Environment (PLANET), the interactive version of Defra's fertiliser recommendation system RB209 (Defra, 2006), providing a farm gate methodology to aid fertiliser recommendations. With many approaches in use, surpluses must be interpreted in the context of the methodology used, respecting the loss pathways already accounted for (Oenema et al., 2005). Whilst the variety of accounting systems and extent of their adoption is encouraging, a uniform and coherent concept for budget calculations at field and farm scale is required (Oborn et al., 2003; Oenema et al., 2003). In addition, farm type and production intensity inherently affects farm efficiency (Brouwer, 1998; Domburg et al., 2000; Lord et al., 2002), meaning surpluses should be interpreted in the context of farm system.

Nutrient balances identify where supply exceeds demand and a nutrient surplus exists. Nutrient loss and enrichment of water bodies is often associated with excessive inputs (e.g. Domburg et al., 1998) and so a connection can be made between nutrient surpluses and potential loss. Corresponding reductions in surpluses and leached losses have been observed for N in the Netherlands (Aarts et al., 2000), Norway (Bechmann et al., 1998), and New Zealand (Power et al., 2002), and between surpluses and river nitrate concentration in England and Wales, although only for grass dominated catchments (Lord et al., 2002). However, the relationship between N surpluses and loss is sensitive to climate, topography, land use history, soil properties and agricultural system, and it is accepted budgets predict only potential loss (Oborn et al., 2003; Oenema et al., 2003).

Potential losses estimated from nutrient surpluses provide an opportunity to assess the effectiveness of mitigation options if one assumes a lower potential loss leads to a lower actual loss. ‘Source’ mitigation methods aim to redress the balance between nutrient supply and demand in order to minimise nutrient loss. Balance inputs can be simply adjusted to simulate the implementation of these methods, and changes are reflected in a reduction in the nutrient surplus. However, budgets are often calculated on an annual time step, meaning ‘timing’ methods which exploit the optimisation of nutrients and the synchronicity between supply and demand in space and time are poorly represented (Oborn et al., 2003). Although less applicable to nitrate loss, transport methods which aim to reduce the risk of nutrient mobilisation and transfer to water bodies have no effect on inputs or outputs and so budgets are insensitive to their implementation.

Despite their inherent sensitivity to source based mitigation few budget based evaluations of mitigation can be found in the literature. Kuipers and Manderloot (1999) offer a rare example, presenting likely surplus reductions induced by a range of mitigation methods, albeit from modelled scenarios. Nutrient budgets are instead more commonly used to track changes in nutrient use (e.g. Kyllingsbaek and Hansen, 2007), increase awareness of nutrient management (Halberg et al., 2005) or adopted in a regulatory context. Farmgate budget based approaches have been utilised in the Netherland and New Zealand to reduce nutrient loadings to water nationally. In the Netherlands farmers faced costly levies where surpluses corresponding with required water quality standards were exceeded (Ondersteinjn et al., 2002; Oenema et al., 2005). However due to background leaching contributions and variation in precipitation, improvements were smaller than expected and the approach replaced by annual fertiliser and manure application limits (Oenema et al., 1998; LNV, 2004; Oenema et al., 2005). In New Zealand the farm budget based model OVERSEER (Wheeler et al., 2003) fulfils a regulatory role, calculating maximum permissible leached losses (Shepherd et al., 2009). Yearly variation in climate is ignored and as a result agreement between measured and modelled estimates of N leaching at the field scale differed by <20% on four dairy farms (Power et al., 2002). The approach is being adopted nationwide and represents the main nutrient support tool in New Zealand (Ledgard et al., 2004).

The availability of input data in farm records and national datasets has allowed budgets to be successfully applied to a wide range of scales from field to national scale (e.g. Withers et al., 2001). They are particularly useful at the farm scale where they are able to confirm compatibility with existing farm systems and engage farmers. Spatial and temporal variability in sources and delivery processes means losses predicted from field scale surpluses should not be upscaled to the whole catchment. Conversely, surpluses calculated using low resolution national datasets should not be extrapolated to individual farms (Domburg et al., 2000).

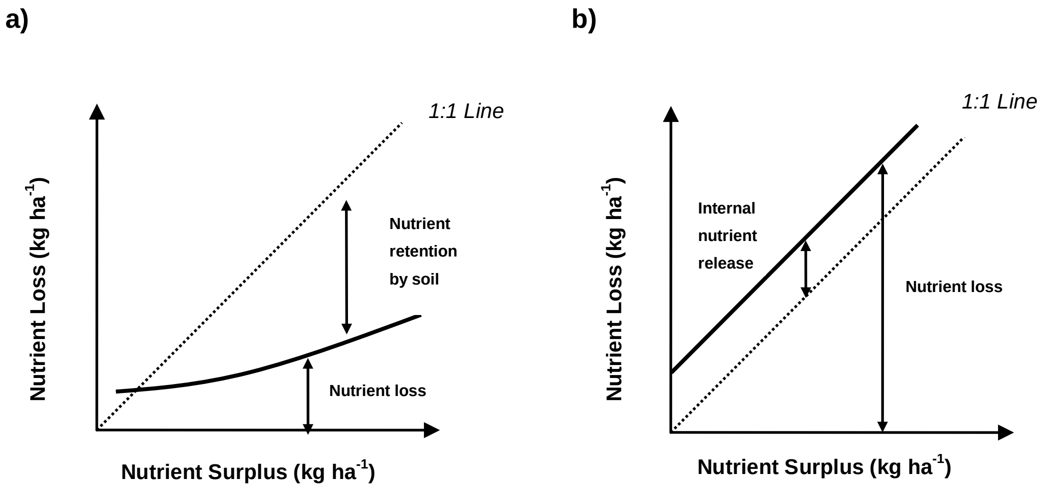


Figure 2-9: The indirect relationship between nutrient surpluses and nutrient loss

Where surpluses are used to estimate potential loss, it is important that uncertainties surrounding the surplus-loss relationship are considered. Budgets do not account for nutrients released from saturated soils (Figure 2-9a) and are insensitive to the retention and subsequent establishment of long-term surpluses (Figure 2-9b). Characterising initial soil nutrient levels and soil properties would go some way to addressing this problem. Budgets assume even distribution of surplus and loss processes. Surplus – loss relationships are therefore only representative of sites where surpluses are evenly distributed and loss processes are proportional to surpluses (Van Beek et al., 2003). Van Beek et al. (2003) also demonstrate nutrient loss occurs at the field scale and intra farm variability exceeds that of inter farm variability suggesting field scale balances are likely to be the best indicator of nutrient loss (Watson and Atkinson, 1999; Van Beek et al., 2003). The N content of the soil is a function of the short-term balance between N mineralisation, immobilisation and crop demands, not the annual accumulated soil N as assumed by soil system balances (Withers and Lord, 2002). Budgets make the assumption that any surplus coincides with drainage. However leaching may occur before crop nutrient demand in the same budget year. To improve the relationship between surpluses and loss, Oborn et al. (2003) and Bechmann et al. (1998) propose budgets should be averaged over a number of years to eliminate the effect of temporal variation in climate and farming practices.

Bias and error introduce further uncertainty into nutrient budget assessments (Oenema et al., 2003). Budgets are an interpretation and simplification of agrosystems which are complex and varied, and methodologies are often inconsistent and assumptions poorly defined (Brouwer, 1998; Oborn et al., 2003; Oenema et al., 2003). Similarly, few common and accepted reference values are available thus limiting applicability as a performance indicator (Oenema et al., 2003). Input data may originate from many sources, at different levels of detail, the error of which is often unknown (Domburg et al., 2000). The N content of manure is often estimated due to heterogeneity in nutrient content and difficulties in obtaining representative samples (Oenema et al., 1998). Uncertainty is therefore greater where large amounts of manure are exported or imported (Oenema et al., 1998) and would be reduced where actual N contents are available (Domburg et al., 2000). Difficulties in conducting representative surveys of grass yield and N offtake in the UK introduces additional uncertainty into grassland soil surface balances (Lord et al., 2002). Farm gate balances are generally considered more accurate than soil surface and soil system balances due to greater certainty in the N content of major fluxes such as feed, fewer sources of input data, and less sampling derived data (Lord et al., 2002). Soil system balances require detail of soil processes which are often estimated to avoid extensive sampling programmes (Watson and Atkinson, 1999). Minimising uncertainty is particularly important at the farm scale where system investigations cannot be replicated. Increasing the availability of actual N content data and ensuring farmers update annual records correctly and completely is therefore especially important. Field scale replicates should be supported by monitoring of soil nutrients to check the accuracy of budgets (Oenema et al., 2003). However, sampling of soils introduces another layer of uncertainty (e.g. Edwards et al., 1997). Since there is difficulty in quantifying the error in budget calculations, conservative assessments and safety factors are recommended (Oenema et al., 2003). However, in doing so confidence intervals may exceed the impact of mitigation.

2.7.3 Modelling

Models aim to characterise and quantify nutrient transport, retention and transformation using empirical equations that describe a physical system. Many nutrient loss models exist (Table 2-2), ranging in complexity from simple empirical applications to comprehensive, fully process driven models. For the purpose of mitigation evaluation, inputs and parameters are adjusted to simulate the implementation of mitigation methods, and predictions of nutrient loss under mitigation and baseline scenarios compared to assess mitigation effectiveness (e.g. Flynn et al., 2002). Empirical models are more commonly used at larger scales (e.g. Johnes et al., 2007), a result of their lower and more readily available input data requirements. But with fewer parameters for adjustment and longer timesteps, simpler models are limited in terms of mitigation sensitivity. Operating at higher temporal resolution and with a wide range of inputs and parameters, process based models are sensitive to a much wider range of mitigation options (e.g. Whitehead et al., 1998; Lord et al., 2007). Higher spatial resolution increases sensitivity to the placement of mitigation and highlights where future mitigation should be targeted (Collins et al., 2007). However with increasing complexity comes larger data requirements and more extensive computation limiting the spatial extent of their application (Wade et al., 2004).

Table 2-2: Examples of N loss models for the evaluation of mitigation effectiveness.

Model	Reference	Model type	Scale of applications	Temporal resolution	Type of mitigation evaluated
Export Coefficients	Johnes, (1996)	Empirical	Catchment - national	Annual	Source[1] and transport[2] (non explicitly)
NGAUGE	Brown et al., (2005a)	Empirical	Field and farm	Monthly	Source[1] and timing[3]
INCA	Whitehead et al., (1998); Wade et al., (2002)	Process	Catchment	Daily	Source[1], timing[3] and transport[2]
NIPPER	Lord et al., (2007)	Process	Field and catchment	Daily	Source[1], timing[3] and transport[2]
SWAT	Arnold et al., (1998)	Process	Catchment - national	Daily	Source[1], timing[3] and transport[2]

[1] Source mitigation aims to redress the balance between nutrient supply and demand (e.g. fertiliser recommendations)

[2] Transport mitigation aims to reduce nutrient mobilisation and transfer (e.g. cover crops and buffer strips)

[3] Timing mitigation ensure temporal synchronicity between nutrient supply and demand (e.g. spring application of manure)

Underpinning the success of model based evaluation is the sensitivity of models to the adjustment of parameters and inputs (Liu et al., 2005), and quantification of the impact of mitigation on these values. Considerable empirical evidence is required to ensure adjustment is objective. However the availability of such data remains limited, especially at the catchment scale and where mitigation targets the delivery of nutrients to water bodies. The development of reduction factors aims to increase standardisation of parameter adjustment but existing values require further refinement (Gitau et al., 2005). Evaluations of mitigation effectiveness demand models with greater predictive sensitivity compared with those used for nutrient loss characterisation (Lord et al., 2007). While extensive validation is therefore essential, the availability of empirical data limits the number of models which have undergone adequate testing. Coupled with uncertainty from parameterisation and input data, model based evaluations are also confounded by structural uncertainty.

3 Introduction to the study area

3.1 Introduction

This chapter aims to introduce the study area, placing it in the context of environmental and agricultural characteristics found in the wider region and indeed country. It also aims to justify why this area was chosen with regard to the aims set out in chapter 1.

3.2 Physical characteristics

3.2.1 Location and size

Investigations were centred on three catchments in south Dorset; Milborne St Andrew and Dewlish (MSA), Empool and Eagle Lodge (Figure 3-2). MSA is situated in the upper Piddle catchment approximately 20km southwest of Blandford Forum. Empool and Eagle Lodge are located in the upper catchment of the River Frome, north and south of Dorchester respectively. However due to their hydrological and geographical proximity Empool and Eagle Lodge are, from here on in, considered as one catchment and referred to as EMEL. EMEL covers an area of approximately 7350ha (73.5km^2) two thirds of which is within the Empool region. MSA is considerably smaller at 4470ha (44.7km^2).

3.2.2 Geology

MSA and EMEL are underlain by cretaceous chalk most of which is unconfined allowing rainfall to percolate to depth (Figure 3-3). Upper Greensand and gault clay deposits are exposed in some valley headwaters, for example in the northwest portion of MSA, whilst Palaeogene sands and clays confine the chalk aquifer in the eastern most part of Empool. Groundwater tends to flow in a southerly or easterly direction, predominately in the upper 50m of the water table where fractures and bedding planes have been enlarged by solution. EMEL and MSA are characterised by high transmissivity which suggests fissure not matrix flow represents the bulk of water movement (Rukin et al., 2008). Leachable N is therefore able to move rapidly through the profile under heavy rain resulting in short term variation in N concentration and a ‘spiky’ N response. The unconfined chalk provides a good groundwater supply, supplying 90% of public water supplies within the Frome and Piddle groundwater monitoring unit. This is much higher than the 35% of water sourced from groundwater observed nationwide.

3.2.3 Hydrology

The permeable nature of MSA and EMEL means the majority of rainfall percolates to underlying chalk aquifers. However water emerges at a number of springs in both catchments, forming small headwater streams of the Frome and Piddle rivers (Figure 3-1 a and b). Once established the River Piddle flows southeast towards Wareham before entering Poole Harbour. The River Frome flows east through Dorchester before entering Poole Harbour at a similar location to that of the Frome. In their middle reaches both rivers display a network of braided channels before developing broad flood plains and marshes in their lower reaches.

Figure 3-1: Typical MSA hydrology a) Groundwater emerging at Warren Farm Spring, b) Dewlish Stream c) Overland flow and soil erosion in MSA

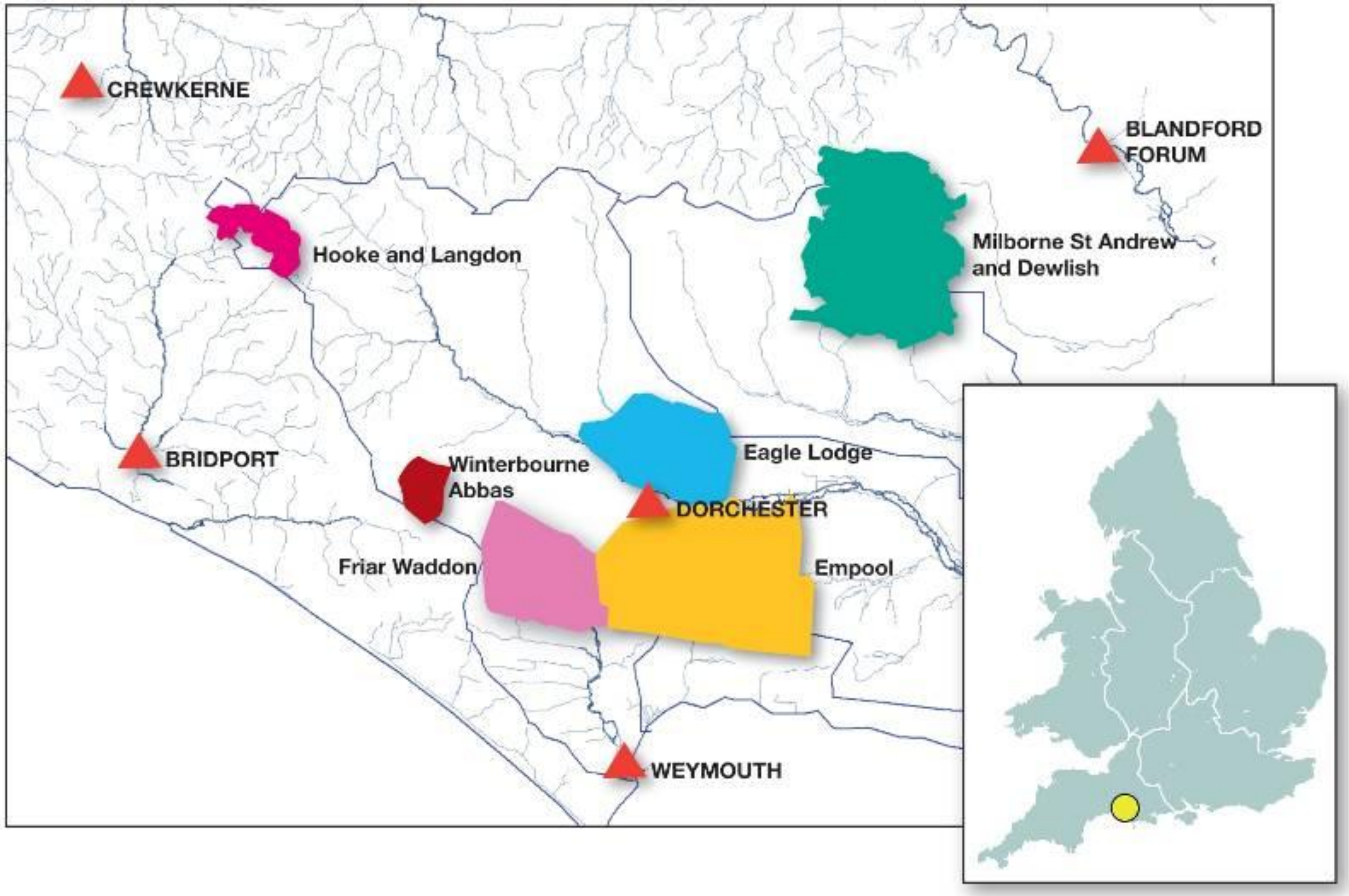


Figure 3-2: Location of Milborne St Andrew and Dewlish (MSA) and Empool and Eagle Lodge (EMEL) (WAgriCo, 2006). Friar Waddon, Hooke and Langdon and Winterbourne Abbas represent additional WAgriCo pilot catchments – see section 3.5.1 for further details.

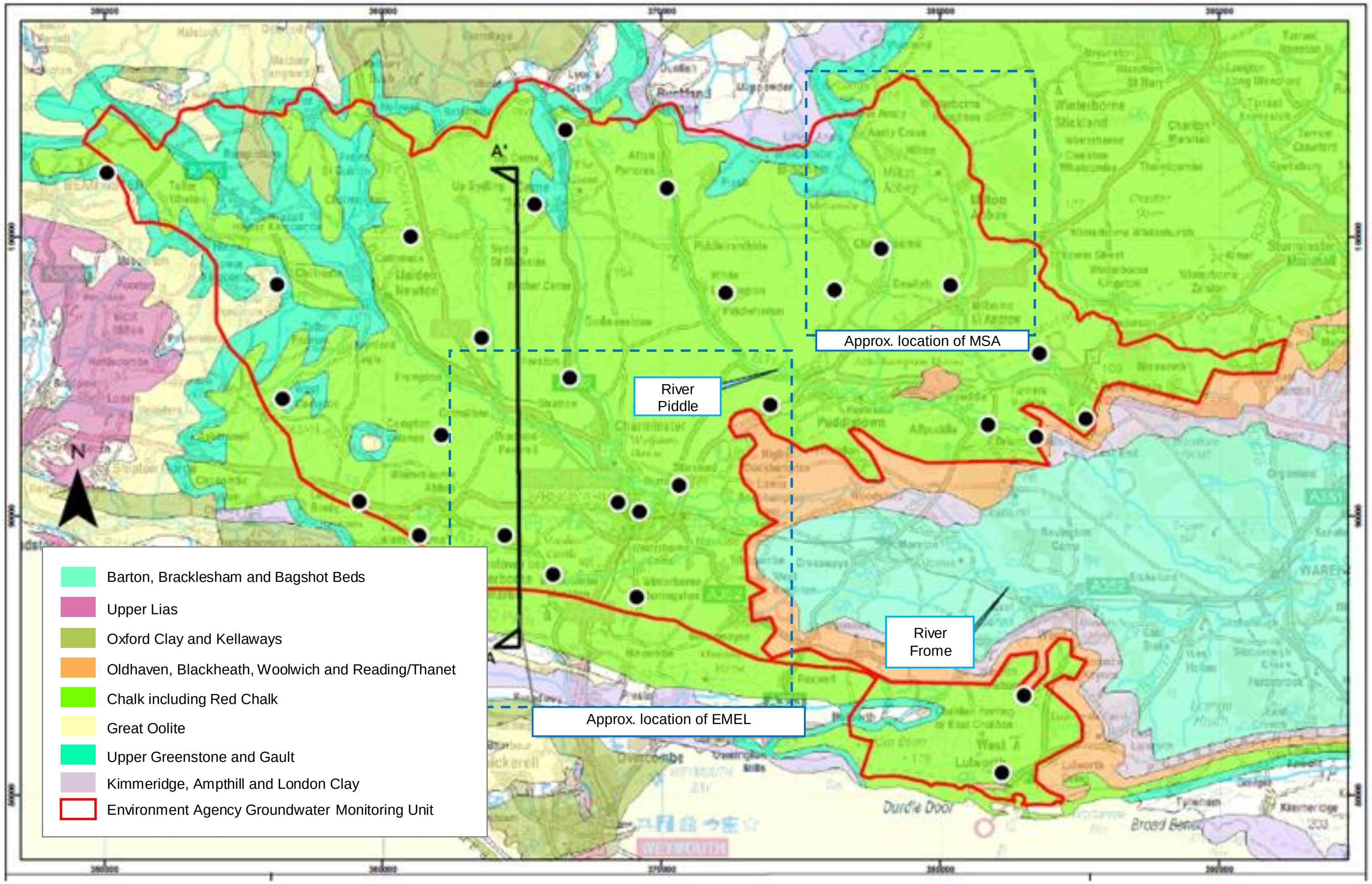


Figure 3-3: Geology of the Piddle (MSA) and Frome (EMEL) catchments (Brown et al., 2005b)

3.2.4 Topography

MSA is located on the edge of the Dorset Downs, with northern parts of the catchment in excess of 200m and a maximum height of 278m. The catchment is dissected by a number of small valleys with steep valley slides which give rise to localised surface run off and soil erosion problems (Figure 3-1c). Small streams flow within these valleys in a southerly direction. In contrast EMEL is lower lying, and characterised by gradual gradients, sloping towards lower lying Dorchester and land to the east of the catchment. However a noticeable ridge runs northwest to southeast in the south of the catchment reaching heights greater than 150m. Figures 3-4 and 3-5 illustrate landscapes observed in each catchment.

3.2.5 Soils

Soils in MSA are typically shallow, well drained and chalky, although there are areas of heavier, clay-influenced soils. Soils on valley sides tend to be well drained and sandy, most commonly over gravel, whilst those on higher land are more calcareous and contain flint. Located further downstream than MSA, EMEL displays sandier and more acidic soils. Closer to Dorchester and to the main river channel soils contain alluvium, exhibiting more clayey characteristics.

3.2.6 Climate

Dorset has a temperate climate characterised by warm summers and mild, wet winters. Temperatures average 8.1 / 1.4˚C and 21.7/11.9˚C (max/min) in January and July respectively (1971-2000 average at Yeovilton Met Office observation station, 40km northwest of the study area) (Met Office, 2010a). Average annual rainfall in the region ranges from around 800mm on the coast to >1000mm over the Dorset Downs (Met Office, 2010b). MSA and EMEL occupy an intermediate position, with annual rainfall averaging 916 and 888mm in MSA and EMEL respectively (average of 1961- 1990 data) (UNESCO, 2007). December and January are the wettest months with monthly rainfall totals c.50% higher than in the spring / summer (Met Office, 2010a).

1) Looking southeast towards Gallows Corner and central MSA

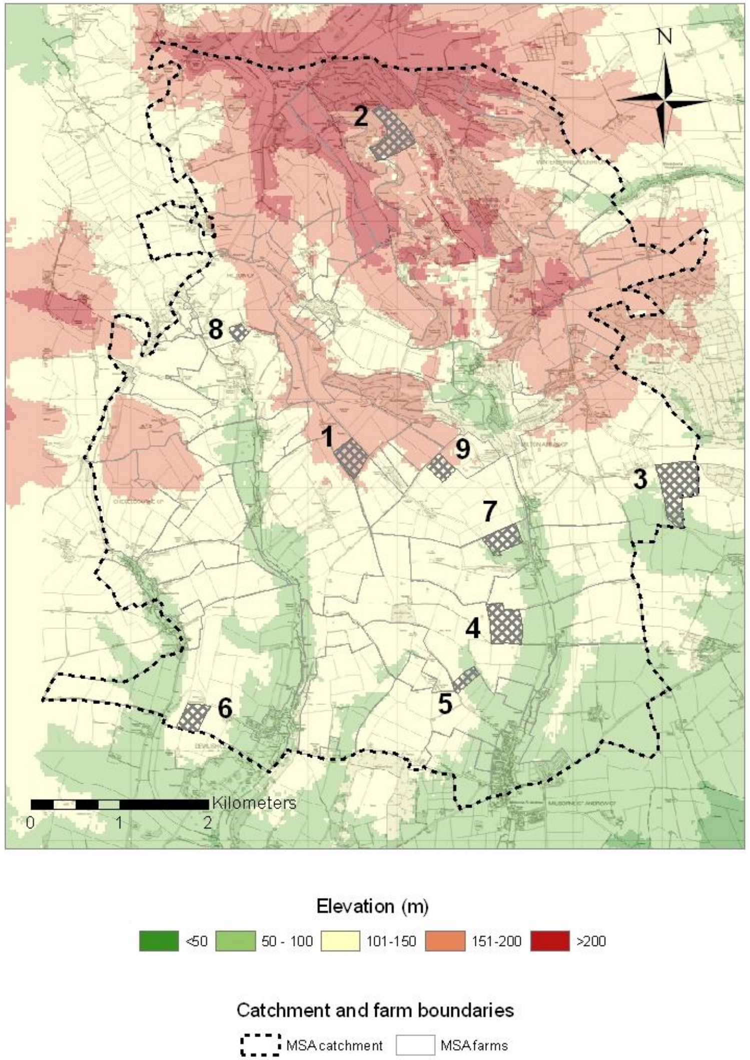
2) Looking northwest from Delcombe farm towards Delcombe Wood

3) View south from Luccombe farm

4a) View northwest from Oat field (Bagber Farm)

4b) Looking northeast towards Little Hewish and Hewish Farms

4c) View southeast from Oat field (Bagber Farm)



5a) Looking northeast towards Bagber Farm from Top Field (Coles farm)

5b) View east from Top field towards Deverel farm

6) Cows grazing on Top O'west (Manor farm) at Whitelands Down

7) View west from Top Hill on Little Hewish Farm

8) Looking Northeast towards Links Plantation on Combe Hill

9) Sheep grazing on Long Close Farm

Figure 3-4: MSA topography and landscapes. Numbers refer to photo locations

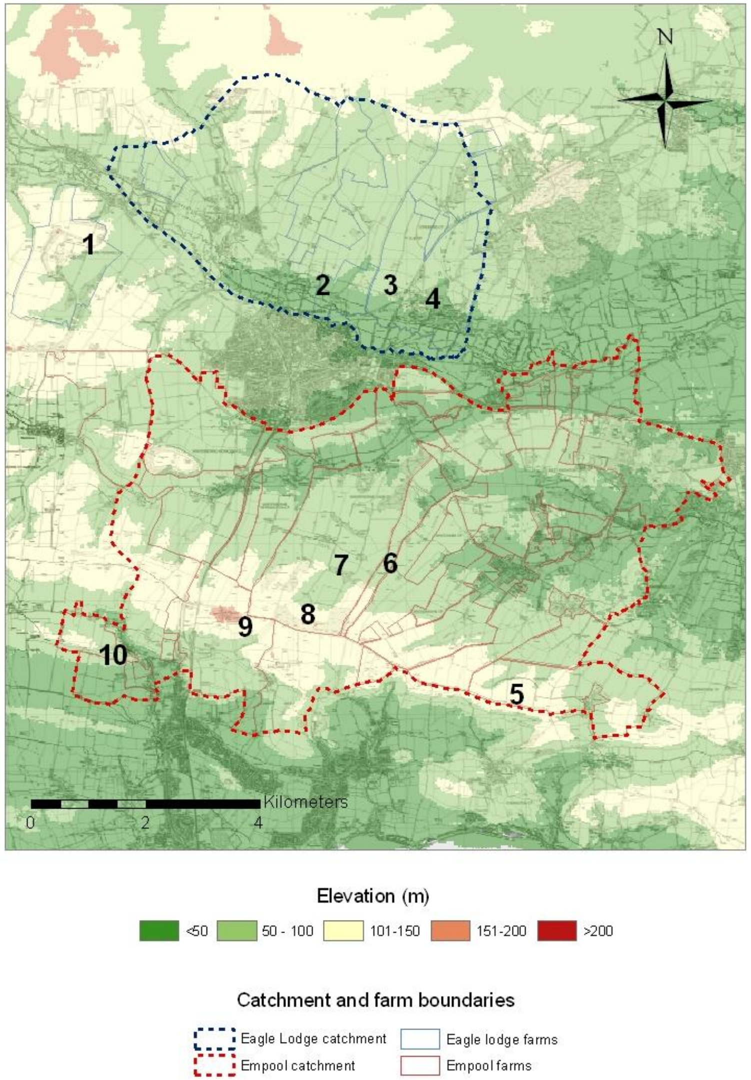
1) Winter Barley at Lower Skippett Farm

2) Sheep grazing at Cokers Frome Farm

3) Pigs at Kingston Maurwood College Farm

4) Sheep at Kingston Maurwood College farm

5) View northwest from Beech farm over organic grass fields



6) Looking south over arable fields on Whitcombe Barn Farm

7) View northeast across Came Estate Farm

8) Came Wood on Came Estate Farm

9) Looking southwest towards Weymouth and Portland from ridge on Higher Ashton Farm

10) Looking northeast towards Dorchester from Higher Ashton farm

Figure 3-5: EMEL topography and landscapes. Numbers refer to photo locations

3.3 Land Use

Land use in the South West (SW) of England (which includes Dorset) is predominately agricultural. In 2008 agriculture covered 1885692ha across some 53718 holdings equating to 90% of the total area; this is considerably higher than across the whole of England where agriculture occupies 70% of land. Permanent Pasture (PP) represents the dominant land use in the south west with crop production utilising c.40% less land than observed nationally (Figure 3-6).

Similar to the wider SW region, land use in MSA and EMEL is predominately agricultural, occupying 86% and 80% land in MSA and EMEL respectively. While EMEL borders the market town of Dorchester, both catchments are sparsely populated with only small villages and isolated farms.

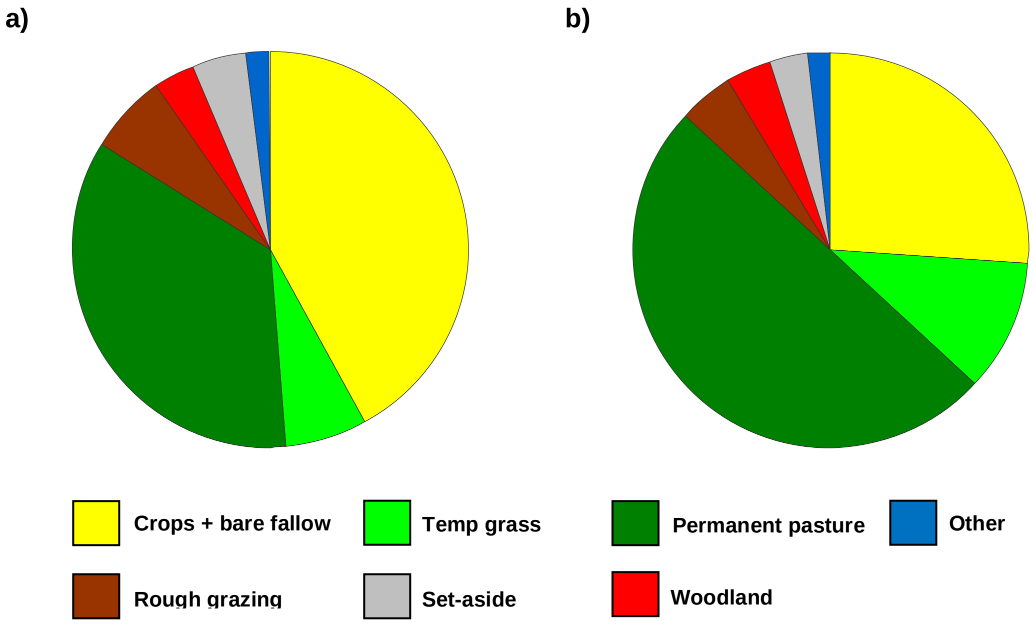


Figure 3-6: Agricultural land use in a) England and b) South West England. Average of 2005 – 2008 June Survey results (Defra, 2008)

3.3.1 Agricultural characteristics

3.3.1.1 Farm type distribution

In accordance with the large area of permanent grass observed in southwest England, cereals farms are less prevalent in the southwest than observed nationally. Conversely a larger proportion of grazing livestock farms are found in the southwest. At both scales lowland grazing livestock farms exceed the number of cereal farms.

However from the data available a different picture emerges in MSA and EMEL where the majority of farms are cereal and mixed respectively. The number of cattle and sheep farms is much lower than observed in the southwest in general (Figure 3-7). However only 62 and 70% of MSA and EMEL were monitored (by area – data availability limited by farmer participation in the WAgriCo project – see section 3.5.1). Given the relative animal husbandry – arable land use in the wider Frome and Piddle catchments (Table 3-1) fewer livestock farms appear to have participated in this project. It is also worth noting that these farm type distributions are based on number of holdings and not farm size. Cereal farms are generally larger than grazing farms.

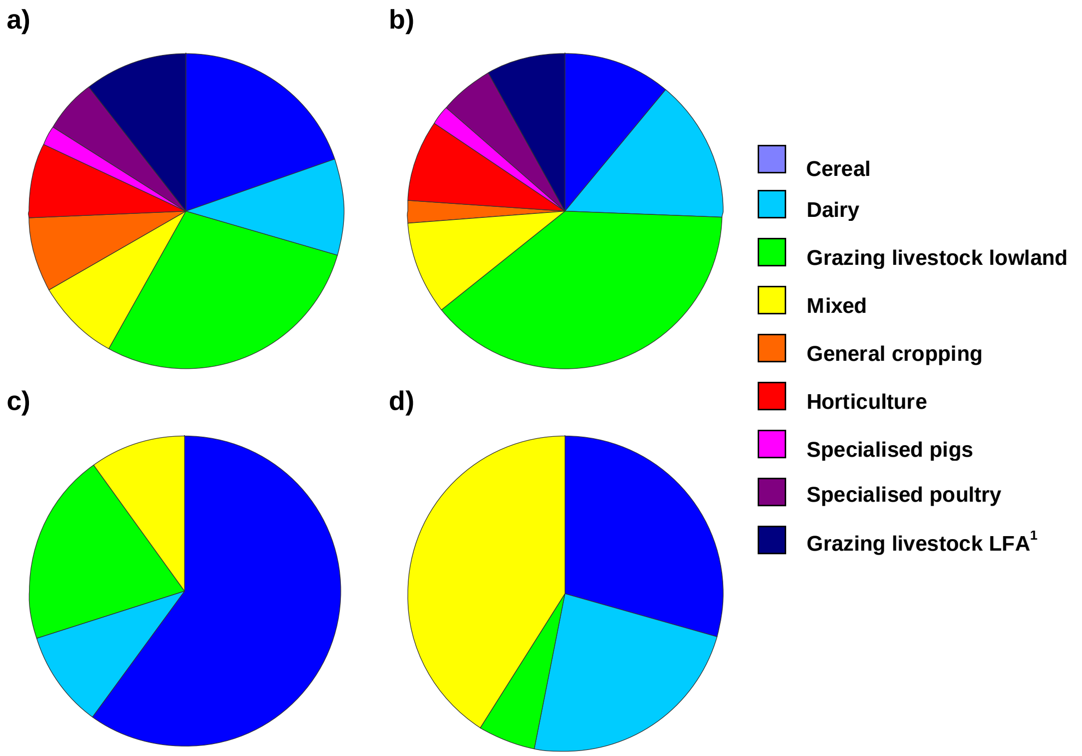


Figure 3-7: Farm type distribution in a) England b) South West England c) MSA d) EMEL. England and SW England results average of 2005 – 2008 June Survey (Defra, 2008). [1]LFA = Less Favoured Area

Comparing the size of farm holdings, the situation in the southwest is very similar to that across the whole of England with the majority of farms less than 5ha (Figure 3-8). However the situation is again very different in MSA and EMEL where the majority of farms are larger than 100ha and the average farm size across both catchments is 213.6ha. Farms are especially large in EMEL where 3 of the 17 monitored farms exceed 500ha.

Table 3-1: Arable and livestock areas in the Frome (EMEL) and Piddle (MSA) river catchments (WAgriCo, 2006).

	Frome (EMEL)	Piddle (MSA)
Area (km^2)	3733	1793
Number of farms	450	200
Animal husbandry [a]	42%	37%
Arable [a]	37%	45%
Other [a]	21%	18%

[a] By area

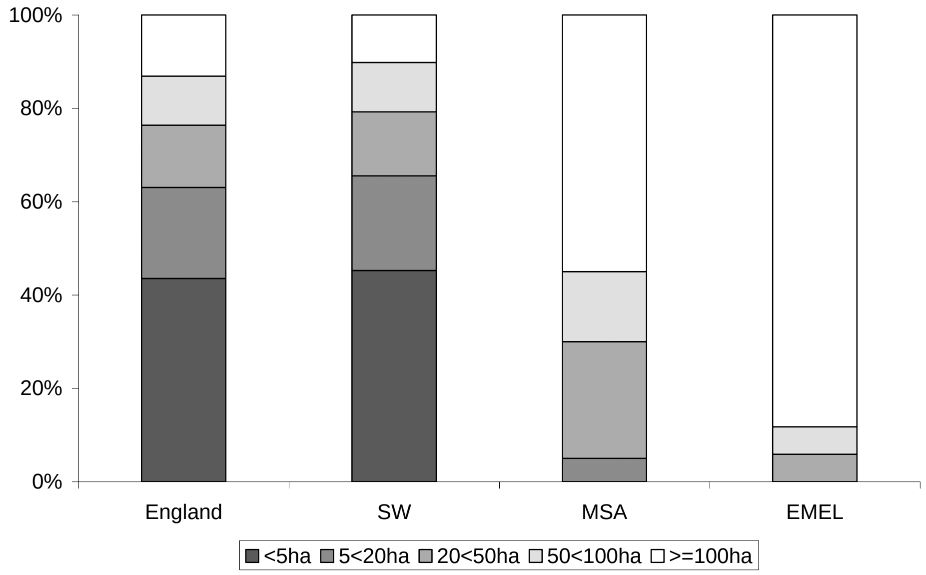


Figure 3-8: Farm size distribution in England, the south west of England, MSA and EMEL. England and SW England results average of 2005 – 2008 June Survey (Defra, 2008)

3.3.1.2 Cropping and stocking

Across MSA / EMEL as a whole, grass represented the predominant crop (c. 40% total catchment area) followed by winter wheat (c.30%) and spring barley (10%). Total grass areas were higher in EMEL than MSA corresponding with higher overall stocking rates in EMEL (0.85 vs. 0.75 LU ha^{-1}). However cropping patterns varied between farm types with maximum grass areas observed on livestock farms (cattle and sheep and dairy), and largest cereal areas on cereal farms (Figure 3-9). Total grass areas reflected stocking densities with larger grass areas on MSA dairies / mixed farms than EMEL mixed / dairy farms due to higher stocking rates. Winter wheat, spring barley and winter oilseed rape are the dominant cereal crops. Maize represents a significant crop on dairy and mixed farms especially in MSA where stocking rates tend to be higher.

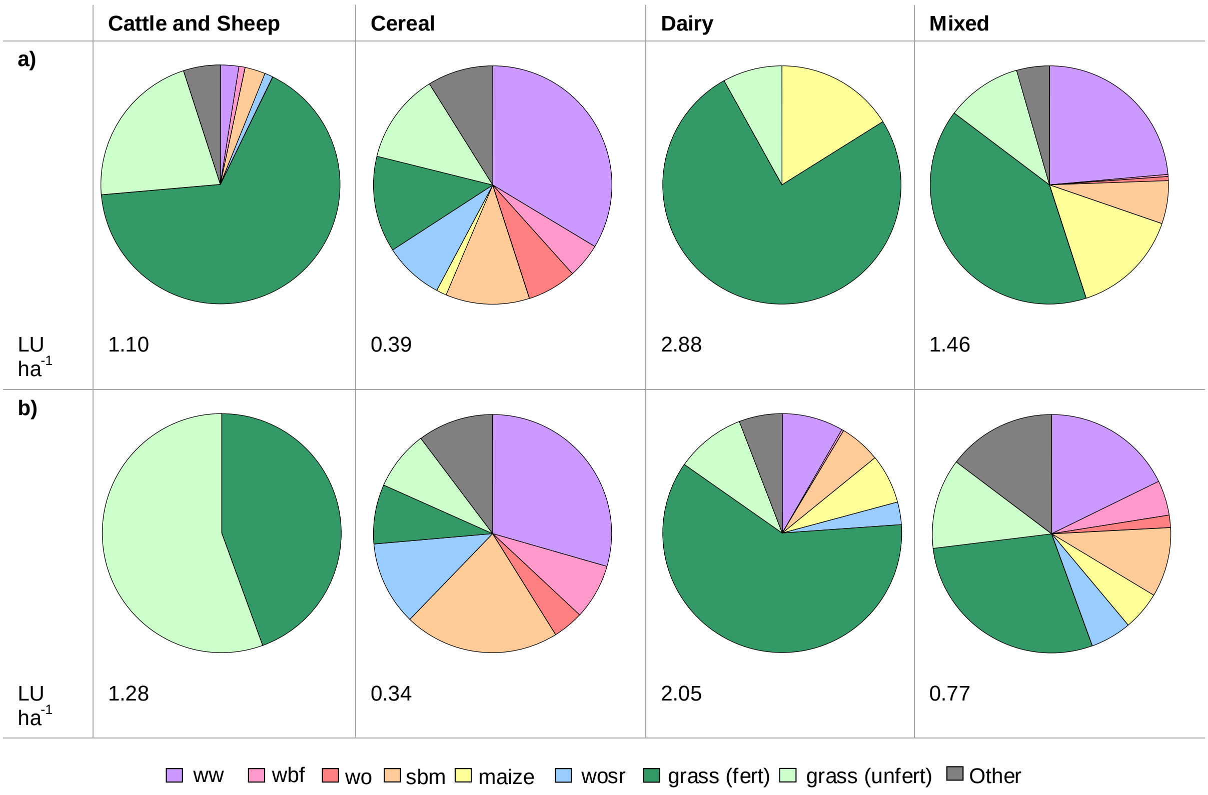


Figure 3-9: Cropping (% total farm average) and stocking density (LU ha^{-1}) in a) MSA and b) EMEL between 2005 and 2008

3.4 Biological and chemical state of the aquatic environment

The proportion of river stretches achieving good or very good environmental standard (according to Environment Agency's general quality assessment) is higher in the southwest than observed nationally. In terms of chemical status 89% of river length is classified as being good or very good compared with 79% nationally, whilst 88% is classified as good or very good biological status compared to 72% nationally (Environment Agency, 2008). However higher standards has limited scope for improvement with larger increases in river stretches achieving good / very good status observed nationwide. Since 1990 the proportion of rivers achieving these higher levels of chemical and biological status has increased by 6 and 14% respectively compared to 24 and 15% nationally. With regard to nitrate, high levels (>6.8mg N l^{-1}) have been observed along 24% of river stretches in the southwest compared to 32% nationally (Environment Agency, 2008).

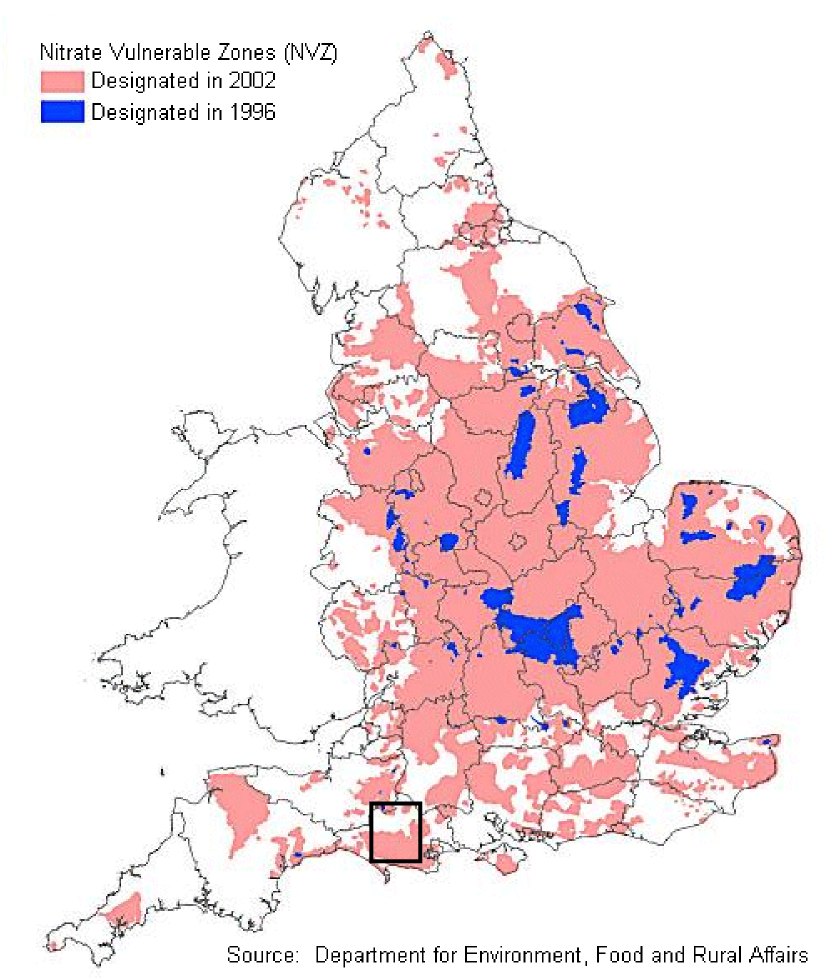
While the status of the aquatic environment is generally better than observed across the rest of England, the recent South West River Basin Management Plan highlights a range of pressures threatening achievement of the environmental objectives set out by the WFD including diffuse pollution from agriculture (nitrate, phosphorus and sediment loss), sewage effluent, contamination from disused mines and unsustainable abstraction (Environment Agency, 2009). As a result only 33% of all surface waterbodies are currently achieving good ecological status and 51% achieving good biological status as defined by the WFD. With respect to groundwater, 84% are of good quantitative status and 64% at good chemical status.

3.4.1 Nitrate pollution

Diffuse nitrate loss from agriculture has resulted in the designation of Nitrate Vulnerable Zones across large parts of England and the southwest (Figure 3-10a). Both MSA and EMEL are located within NVZs, a result of high nitrate concentrations in groundwater and surface water, and eutrophic surface waters (Figure 3-10b). Farmers in both catchments are required to adhere to Action Programmes concerning the time and quantity of N applied especially as manure.

While the proportion of river stretches with high nitrate concentrations has decreased in the southwest as a whole, upward trends have been observed in groundwater (Figure 3-11). Concentrations at public water supply boreholes in EMEL and MSA have increased by around 1 and 3mg N l^{-1} over the last 20 / 30 years respectively. On a number of occasions concentrations in MSA exceeded the drinking water standard of 11.3mg N l^{-1}. As a result Wessex Water (the local water company) has been forced to blend high nitrate waters with lower nitrate waters and build new nitrate removal plants. In an effort to explore more cost effective approaches, Wessex Water began trialling catchment management initiatives for the control of groundwater nitrate concentrations in 2006.

a)



b)

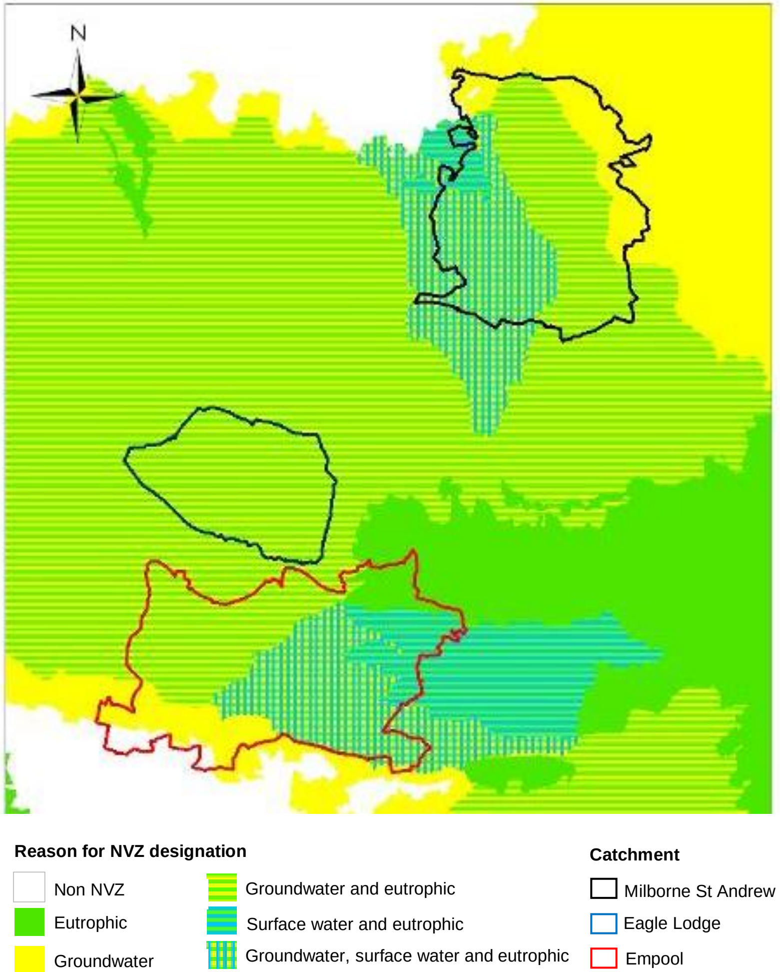


Figure 3-10: a) NVZs in England and b) NVZs close to MSA and EMEL with reasons for designation (Defra, 2010a; Environment Agency, 2010). Highlighted area in a) shows approximate location of area in b).

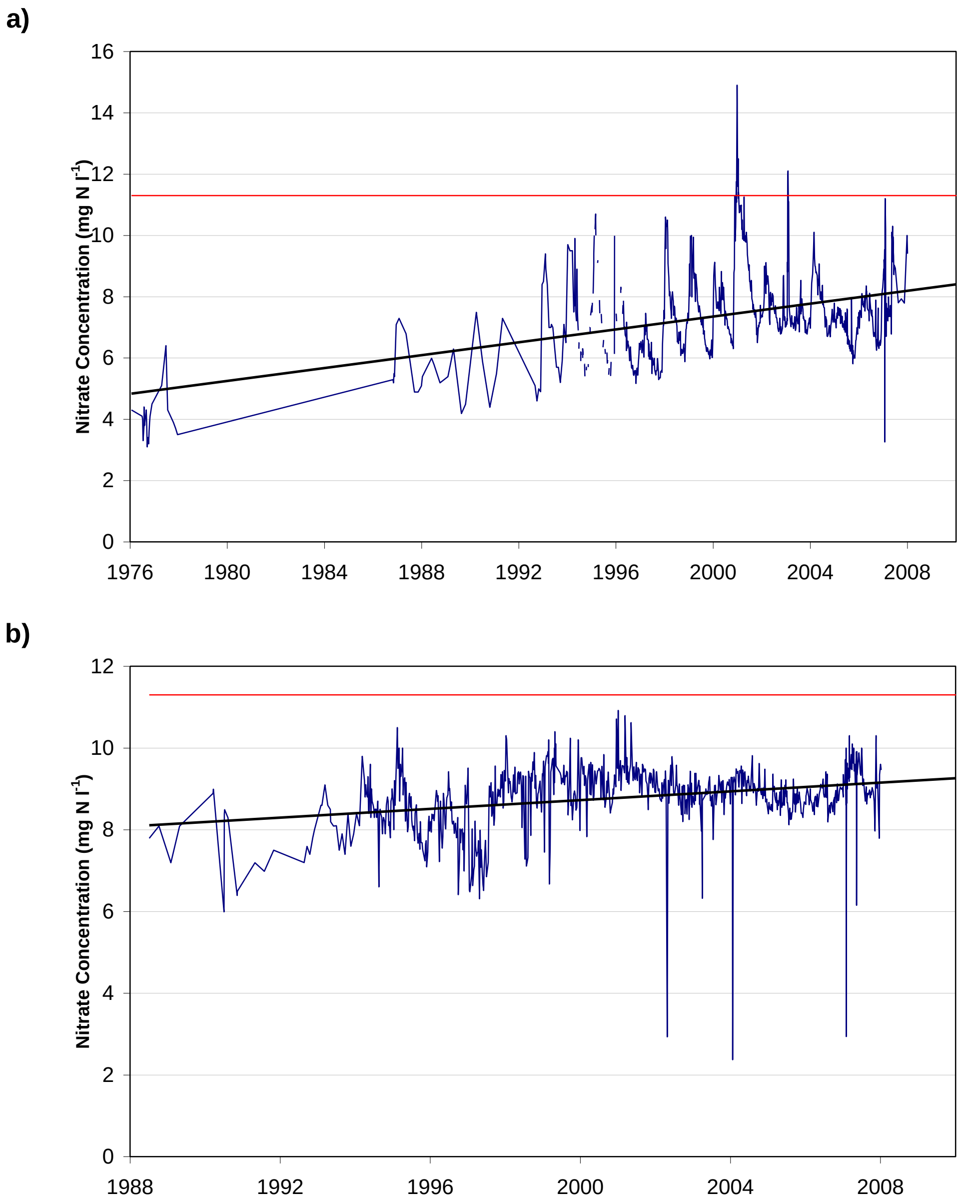


Figure 3-11: Increasing nitrate concentrations (mg N l^{-1}) at public water supply boreholes in a) Milborne St Andrew (MSA) between 1976 and 2008 and b) Empool between 1988 and 2008. In MSA the 11.3mg Nl^{-1} drinking water standard was exceeded on a number of occasions during the winters of 2001/2 and 2003/4.

3.5 Agri-environmental schemes

3.5.1 WAgriCo

Due to rising nitrate concentrations and exceedances of the drinking water standard MSA and EMEL were selected as pilot catchments in the EU Life funded Water Resources management in Co-operation with Agriculture project (WAgriCo). WAgriCo aimed to reduce diffuse nitrate loss to groundwater through collaboration between agricultural and water resource management. The project aimed to develop programmes of measures (i.e. mitigation) to be implemented at the farm level to reduce diffuse nitrate loss from agricultural land. Emphasis was placed on farmer engagement to demonstrate the effectiveness of a supportive approach for the sustainable achievement of environmental objectives as set out by the WFD.

Adoption of MSA and EMEL as study sites facilitated assessment of mitigation implemented in a practical context as part of the WAgriCo project. Farmer involvement made field and farm level data available for the calculation of nutrient budgets, whilst a network of monitoring activities supported measurement based assessment. Involvement in the wider WAgriCo project provided an opportunity to disseminate measurement and nutrient budget results back to farmers to assess the suitability of different assessment methods to different end users, and to appreciate the wider agricultural realities and attitudes associated with diffuse N pollution.

Farmers participated in the WAgriCo project on a voluntary basis but were reimbursed / rewarded for their time and additional costs incurred, typically on a £/ha basis. Following an initial characterisation of the pollution pressures faced in each catchment, a range of relevant, low cost mitigation methods were identified (Table 3-2). A number of more innovative and / or co-operative measures namely fertiliser calibration and ‘efficiency’ were also included. With the help and advice of catchment advisors, farmers selected mitigation most applicable and acceptable to their farms. Regular farmer events, one on one meetings and newsletters were held / produced to offer addition support to farmers and to disseminate results.

Farmer involvement dictated the availability of farm data and hence the monitored area within each catchment. A total of 37 farms were actively involved in the project covering 62% of MSA and 70% of EMEL (by area). Details of farms participating are summarised in Table 3-3. The level of involvement varied between farms; some offered farm data, others allowed soil and water on their land to be sampled whilst others implemented mitigation methods. Table 3-4 identifies mitigation uptake and the classification of farm mitigation levels. Farms adopting only lower level mitigation (fertiliser recommendations and manure management plans) were referred to as GAP farms whilst those adopting higher level mitigation were classified as EGAP farms. Due to its target orientated implementation ‘efficiency’ was excluded from the farm classification process. In most cases adoption of the efficiency measure had no direct impact on farm practice. Figure 3-12 shows the degree of catchment coverage and the spatial distribution of farm types and mitigation level.

Table 3-2: WAgriCo mitigation options and associated farmer support

Mitigation method	Tier / code	Description	Support / co-operation
Fertiliser recommendations	GAP1	▪ Use a recognised fertiliser recommendation system (e.g. RB209, PLANET) to plan fertiliser applications to all crops. ▪ Do not exceed optimum recommended rates. ▪ Time fertiliser applications to minimise the risk of loss of nutrients. ▪ Take full account of manure inputs when planning mineral fertiliser applications.	Provision of nutrient management plan and necessary advice In field sampling Farmer feedback on recommendations
Manure management plans	GAP2	▪ Follow NVZ rules for manure timing, i.e.: ○ Do not apply manure to fields at times when there is a high risk of surface run-off; e.g. in winter when soils are saturated or frozen hard or when heavy rain is expected in the next few days. ○ Do not apply manure to fields at times when there is a high risk of rapid percolation to field drains; e.g. in winter and spring when soils are wet or in summer when soils are dry and cracked over drains. ○ Do not apply manure to fields late in the growing season (i.e. autumn/early winter) or when there is no crop to utilise the added N.	Analysis of manure N content Advice on manure management Calibration of manure spreader In field measurements Farmer feedback
Cover crops	EGAP1	▪ Establish a cover crop immediately post-harvest or, at the latest, by mid-September. ▪ Alternatively, undersow spring crops with a cover crop that will be in place to take up nutrients and provide vegetation cover once the spring crop has been harvested.	Farmer feedback
Fertiliser calibration	EGAP2	▪ Correction of uneven fertiliser sprayers to ensure correct and even applications of fertiliser.	
Delayed manure application	EGAP3	▪ Delaying the application of manure from autumn to spring to make best use of manure N (less is leached and more can be utilised by the established winter crops.	
'Efficiency'	EGAP4	A target orientated mitigation approach aimed at improving nutrient management through improving farm efficiency. Based on an approach developed by German partners in the WAgriCo project which affords farmers greater control in selecting mitigation options and best practices which fit best within their farm. Farms were rewarded for improvements in their efficiency. Calculations of efficiency in chapter 6 were therefore 2 fold; to assess mitigation performance and to reward improved nutrient management.	Farm data required Workshop on how to improve efficiency Farmer feedback

Table 3-3: Farms participating in the WAgriCo project - details of farm type, size (ha) and level of involvement

Catchment	Farm type	No. farms participating	Average size (ha)	SE (ha)	Farms adopting mitigation	Farms adopting mitigation + 4years data
MSA	Cattle and sheep	4	73.80	26.73	3	3
	Cereal	12	144.65	30.26	11	9
	Dairy	2	124.87	93.77	2	2
	Mixed	2	237.67	21.78	2	2
	Sub total	*20*	*137.80*	*22.08*	*18*	*16*
EMEL	Cattle and sheep	1	65.10	0.00	1	1
	Cereal	5	230.79	52.30	5	5
	Dairy	4	370.67	201.75	3	2
	Mixed	7	349.57	63.44	7	7
	Subtotal	*17*	*302.87*	*55.19*	*16*	*15*
TOTAL		**37**	**213.64**	**30.80**	**34**	**31**

Table 3-4: Mitigation uptake in MSA and EMEL, and associated farm mitigation classifications.

		Fertiliser recommendations *GAP1*	Manure Management Plan *GAP2*	Cover crops *EGAP1*	Fertiliser Calibration *EGAP2*	Delayed manure application *EGAP3*	Farm 'efficiency' *EGAP4*	'GAP' farms[A]	'EGAP' farms[b]	No mitigation
MSA	No. farms	18	12	6	8	1	14	9	9	2
	% farms	*90*	*60*	*30*	*40*	*5*	*70*	45	45	10
	Area - ha	2651	1975	180.49[c]	1557	236[d]	2291	845.11	1806.33	104.66
	% area	*96*	*72*	*7*	*56*	*9*	*83*	30.66	65.54	3.80
EMEL	No. farms	16	10	9	10	0	15	3	13	1
	% farms	*94*	*59*	*53*	*59*	*0*	*88*	18	76	6
	Area - ha	5118	3044	298.0[c]	3702	0	5052	624.91	4492.66	31.14
	% area	*99*	*59*	*6*	*72*	*0*	*98*	12.14	87.26	0.60
TOTAL	No. farms	34	22	15	18	1	29	12	22	3
	% farms	*92*	*59*	*41*	*49*	*3*	*78*	32	59	8
	Area - ha	7769	5019	479	5259	236	7343	1470.02	6298.99	135.80
	% area	*98*	*63*	*6*	*67*	*3*	*93*	18.60	79.69	1.72

[a] Farms adopting lower tier mitigation i.e. fertiliser recommendations and manure management plans

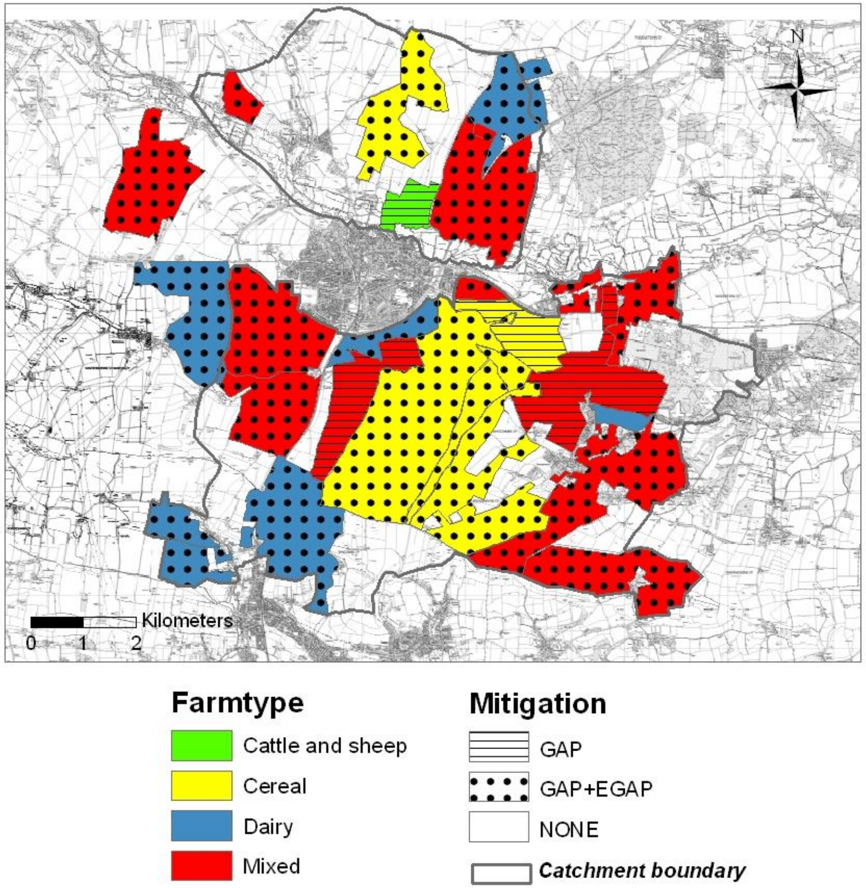
[b] Farms adopting higher at least one higher tier mitigation method in addition to at least one lower tier option. Classification excluded the efficiency measure (EGAP4) on the grounds that it is target orientated not action orientated.

[c] Area covered by cover crops in 2008

[d] Delayed manure applications applicable to winter cereal crops only which amounted to 89ha.

NOTE: Agreement to mitigation methods did not guarantee that options were fully implemented or that any change resulted from their implementation. For example fertiliser sprayer calibration may have already been satisfactory and fertiliser recommendations adhered to.

a)



b)

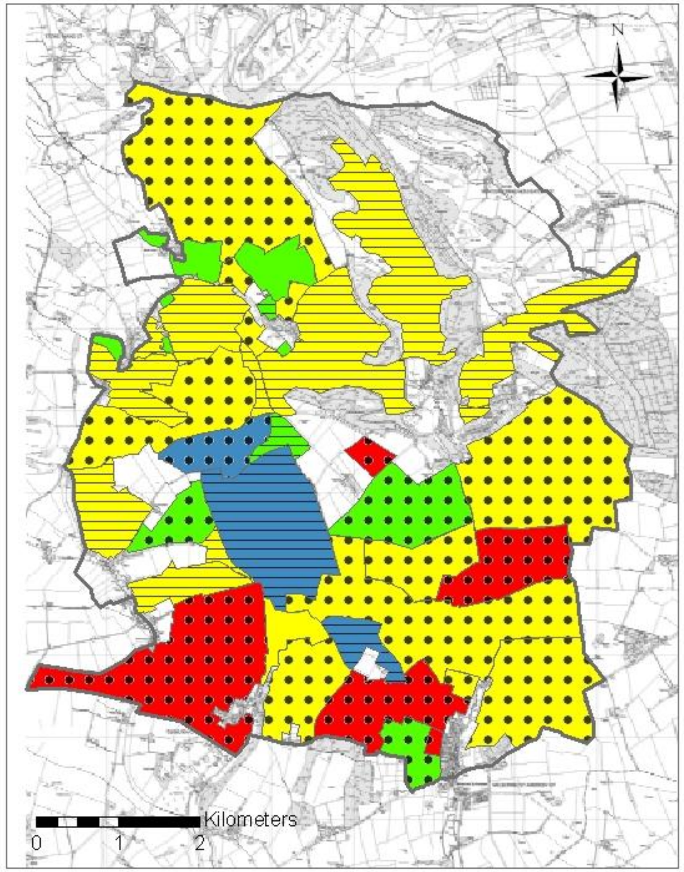


Figure 3-12: Spatial distribution of farms participating in the WAgriCo project in a) EMEL and b) MSA with associated details of farm type and level of mitigation. See Table 3-4 for explanation of mitigation codes.

4 Using field and catchment scale measurement to evaluate mitigation effectiveness

4.1 Introduction

4.1.1 Measurement as an evaluator of mitigation effectiveness

4.1.1.1 Why measurement?

Quantification and characterisation of changes in environmental and ecological state, achieved through measurement of soil, water and biological surveys represent the most direct means of evaluating the impact of mitigation on N loss. Results reflect both agricultural practice and environmental condition, superimposing physical, chemical and biological processes and properties upon land use and nutrient management. Accordingly, measurement approaches have been identified as agri-environmental indicators by Schroder et al., (2004) and Langeveld et al., (2007). Although the extent to which measurement captures external variables depends on the scale and approach adopted, measurement provides an integrated response (Langeveld et al., 2007), and one that is of direct relevance to current legislation.

4.1.1.2 Measurement approaches

The availability of accurate, reliable and inexpensive methods for the measurement of environmental variables is fundamental to the success of studies aimed at reducing N loss (Shepherd et al., 1993). Numerous approaches exist, applicable at a range of scales, although no single technique is considered suitable for all situations (Table 4-1); choice is often constrained by soil type, geology, resources and timescales.

4.1.1.3 Field scale measurement

At the field scale, measurement of soil mineral nitrogen (SMN) and the N concentration of soil leachate collected by porous pots (PP) are the most widely used approaches. SMN reflects the balance of inputs and outputs of mineral N in the soil, quantifying the total crop available N (ammonium + nitrate N). Given the inherent solubility of this N fraction, SMN represents the amount of N at risk of loss to the environment. Porous pots facilitate collection and determination of the nitrate concentration of soil water leaching below crop rooting depth. In doing so field drainage is superimposed upon soil N status.

Table 4-1: Examples of measurement approaches

Method	Details of approach	Example	Advantages	Disadvantages
Soil Mineral Nitrogen (SMN)	Determination of soil mineral nitrogen content (nitrate + ammonium) in soil cores.	Webb et al., (1997); Beaudoin et al., (2005)	Useful for characterising soil N status (Goulding, 2000) Indicator of leaching potential (Shepherd et al., 1993)	Not dynamic (Goulding, 2000) Not a direct measure of leached N (Shepherd et al., 1993)
Porous Pots (PP)	Extraction of soil water through establishment of a pressure gradient between the soil and the inside of a porous ceramic cup.	Vos and van der Putten, (2004); Webb et al., (2004); Sieling and Kage, (2006)	Effective in free draining, structureless soils (Goulding, 2000).	Unsuitable in structured clays (Catt et al., 1998; Goulding et al., 2000) and chalk (Williams and Lord, 1997).
Lysimeters	Collection of water draining from an isolated block of soil.	Catt et al., (1998); Shepherd and Webb, (1999); Goulding et al., (2000)	Measures amount of drainage in addition to concentration of drained water.	Expensive and limited in their applicability (Shepherd et al., 1993). Disturbed soil structure and artificial flow at base (loss of suction) and edges (Armstrong and Burt, 1993; Shepherd et al., 1993).
Isotopic studies	Evaluation of transport pathways using labelled 15N.	Macdonald et al., (1997)	Facilitates comprehensive studies of N fate and transport.	Expensive and complex (Goulding, 2000)
Groundwater / surface water monitoring	Direct sampling of waterbodies	Kronvang et al., (2005); Zwart et al., (2008); Howden et al., (2009)	Maximum integration of catchment conditions. Legislative relevance	Expensive, resource demanding, complex interpretation, time lags / catchment buffering.
Overland flow / surface water collection	Construction of 'run off' facilities / monitoring of field drains / selected fields via flumes and v notch weirs.	Smith et al., (2001); Lord et al., (2007)	Useful where soils are poorly drained and overland flow represents a major N flux.	Expensive and resource demanding

SMN and PP have been widely used to assess of the effect of mitigation on N loss and evaluate responses to agri-environmental policy. Numerous studies have investigated the impact of individual mitigation methods such as cover crops (Beckwith et al., 1998; Shepherd, 1999; Vos and van der Putten, 2004), reduced fertiliser applications (Johnson and Smith, 1996; Johnson et al., 1997 and 2002) and delayed manure application (Smith et al., 2002; Sieling and Kage, 2006) via field scale SMN and PP sampling. A number of these experiments were conducted across whole rotations, effectively offering a farm scale evaluation of mitigation effectiveness (e.g. Johnson et al., 2002). SMN and PP based assessments have also been conducted across whole catchments; their relatively low cost facilitating larger scale applications. In England and Wales SMN and PP sampling was employed to assess the effectiveness of the Nitrate Vulnerable Zone (NVZ) Action Programmes in 16 pilot catchments (Lord et al., 2007), and PPs to evaluate the impact of 'Nitrate Sensitive Areas' in 32 catchments prior to the designation of NVZs (Silgram et al., 2005). PPs are especially useful where catchments are groundwater dominated, capturing changes in the nitrate concentration of water draining to vulnerable aquifers but without the timelags associated with groundwater sampling.

4.1.1.4 Catchment scale measurement

Although loss occurs at the field scale, consequences become apparent at the catchment scale (Withers and Lord, 2002). While field scale measurements are effective in capturing change in the soil N balance and losses from the soil root zone, dilution (from both discharge and non agricultural areas), in transit / in stream transformations, and point source contributions supports measurement at the catchment scale (Schroder et al., 2004; Bechmann et al., 2008; Hutchins et al., 2009). Although responses to change (both environmental and anthropogenic) are longer and reflect a wide array of complex interactions, quantification of surface / ground quality represents the third measurement based agri-environmental indicator identified by Schroder et al., (2004) and Langeveld et al., (2007) (after SMN and PP). With waterbodies the end point of legislative targets, such factors must be considered if measurement is to provide assurance that water quality standards are adhered to.

Time and resources limit the number of comprehensive catchment scale monitoring networks, however numerous studies have been conducted investigating nutrient mobility at the river basin scale both in surface water catchments (Heathwaite et al., 2005b) and in groundwater systems (Howden and Burt, 2008; Howden et al., 2009). Such investigations aim to better understand the complex interactions, transformations and losses occurring at the catchment scale. Long term temporal variability has been investigated to better understand the reasons and timelags associated with groundwater systems; long transit times mean that observed concentrations are often a relic of historic land use and management (Vinten and Dunn, 2001; Limbrick, 2003). In addition, large scale, continual monitoring conducted by national governments provides quality assurance and highlights vulnerable areas with respect to water quality; routine monitoring supported the designation of Nitrate Vulnerable Zones (NVZ) in England and Wales through the identification of waterbodies exceeding or at risk of exceeding the drinking water standard of 11.3mg N l^{-1}. With reference to evaluations of mitigation effectiveness, cost and timelags limit the applicability of catchment scale measurement.

4.1.1.5 Field vs. catchment scale measurement

While both field (SMN and PP) and catchment scale measurement have the potential to provide useful evaluations of mitigation effectiveness, limitations and uncertainty are attached to all three approaches. While SMN provides a rapid result, measurements quantify potential not actual loss, with actual loss dependant on the balance of mineralisation and immobilisation, and the amount of drainage to mobilise leachable N. Porous pots capture actual loss with minimal timelags, but their applicability is soil type dependant, and alike to SMN, field scale losses do not reflect directly what is observed in water bodies. Capturing catchment responses through catchment measurement is costly though, requiring long time series and complicated by the many factors integrated in catchment scale measurement. With respect to the attributes of an ‘effective agri-environmental indicator’ indentified by Schroder et al., (2004) and Langeveld et al., (2007), SMN and PP measurements are more attributable to the management of individual farms and related more directly to farm practice than environmental condition. However with waterbodies status the driver of changes in farm practice, and water companies not farmers accountable for exceedances of the drinking water standard, it could be argued that environmental variables are equally important.

To exploit the benefits of each approach and minimise uncertainty, some studies have incorporated a range of approaches. SMN, PP, overland flow and SW / GW sampling were all employed to evaluate change to nutrient policy catchment wide in Norway, Estonia, Denmark and Finland (Granlund et al., 2005; Iital et al., 2005; Kronvang et al., 2005; Bechmann et al., 2008). Typically, confirmation of effectiveness at the field scale precedes wider implementation in national policy and subsequent evaluation at the catchment scale. Ultimately the aim of the study, the availability of resources and the nature of the site is likely to determine which approach is most appropriate.

4.1.1.6 Measurement as an evaluator of mitigation effectiveness in the current study

Legislative relevance and sensitivity to environmental condition and agricultural practice means measurement represents the most direct method of mitigation evaluation. Owing to the provision of cost effective results over relatively short term time scales, field scale SMN / PP evaluations of mitigation effectiveness are relatively common in the literature. However most studies focus on a limited number of fields / farms, evaluate the maximum benefit of mitigation on an unrepresentative 'treatment' approach, and are rarely supported by measurement at the catchment scale, or indeed nutrient budget approaches. There remains a need to evaluate mitigation adopted in a practical context. The implementation of mitigation in MSA and EMEL provides an opportunity to explore whether SMN / PP measurements capture responses to voluntary WAgriCo mitigation in the short term (2 years).

At the catchment scale an absence of groundwater measurement based mitigation evaluations reflects in part the long timescales required to capture change. However where the underlying rock is highly fissured and leachable N able to move rapidly to depth, responses to mitigation may be observed in groundwater in the short term. With the aquifers beneath MSA and EMEL displaying short term variation in N concentration (Rukin et al., 2008), the catchments are good candidates for investigation of short term responses to mitigation in groundwater. The implementation of mitigation above these aquifers provides an opportunity to investigate whether catchment measurement could, in some catchments, prove a viable means of short term mitigation evaluation.

While previous studies have utilised field and / or catchment scale measurement for evaluations of mitigation effectiveness, few compare the different approaches. The short comings of field scale measurements in relation to catchment scale responses, and the extent to which field scale responses have so far transpired in catchment scale measurements have seldom been considered. Given the opportunity to conduct measurement and evaluations at both scales and through a range of methods, this study provides an opportunity to compare such responses and to investigate the effect of measurement approach choice on evaluations of mitigation effectiveness.

4.1.1.7 Objectives

To address the aims and hypothesis presented in chapter 1, the following objectives were identified:

1. To explore spatial and temporal variability in field and catchment scale measurements.
2. To investigate the impact of mitigation on measurements by:
    a. Comparison of field scale measurements with and without / before and after the implementation of mitigation.
    b. Analysis of time series of catchment scale measurement throughout the WAgriCo project (interjected by the implementation of mitigation catchment wide).
3. To compare field and catchment scale responses to mitigation as captured by field and catchment scale measurement.

4.2 Methodology

4.2.1 Field scale

At the field scale soil mineral nitrogen (SMN) and porous pots (PP) sampling were chosen as N loss measurement approaches due to their relative low cost, simplicity yet sensitivity to changes in agricultural practice and environmental conditions (see section 4.1.1.2). Details of SMN / PP field selection can be found in section 4.2.1.3.

4.2.1.1 Soil Mineral Nitrogen (SMN)

Sampling approach

SMN was measured in both 2007 and 2008 harvest years, before and after the implementation of mitigation. Samples were taken in late autumn / early winter

(November / December) as the soil returned to field capacity prior to the onset of drainage, and again in early spring (February / March) before fertiliser / manure and significant spring mineralisation inputs. Soil was collected at 10 locations per field following a 'W' shaped sampling pattern to account for spatial heterogeneity, avoiding atypical areas such as headlands, tramlines and manure heaps. Soil samples were obtained using a Hydrocare MCL2 Sampler which utilises a high frequency hammer to drive an open slot auger to depth. Samples were taken in 30cm increments to a depth of 90cm; the outer layer of soil was removed to prevent contamination between depths. Replicate samples were bulked together to give one sample at each depth per field, and chilled to minimise mineralisation. Total mineral N content was determined using an ion specific electrode for nitrate using a water / calcium sulphate extractant digest.

To account for N immobilised in plant material, spring SMN sampling was accompanied by crop sampling. Following the random placement of a 25cm^2 quadrat, all above ground plant material was removed, taking care to avoid soil and livestock manure. This was repeated 6 times before being bulked to give one sample per field. Total N content was determined using the Kjeldahl method.

SMN calculations

Analysis determined the total concentration of mineral nitrogen (ammonium + nitrate N) at each depth on a mg kg^{-1} basis. In order to calculate the quantity of SMN on an area basis (per ha) (accounting for soil volume and bulk density), the following calculations were performed:

1. The weight of soil within the sampled layer was calculated using a standard dry bulk density (1.33 kg m^{-3} for mineral soils) and the known thickness of the sampled layer:

   Weight of soil (t ha^{-1}) = 100 x thickness of sampled layer (cm) x bulk density (kg m^{-3})

2. The SMN (kgha^{-1}) for each depth was then calculated from the mineral nitrogen concentrations and soil weight:

   SMN (kg ha^{-1}) = (Weight of soil (t ha^{-1}) x SMN concentration (mg kg^{-1})) / 1000

3. The total SMN for the soil profile is then the sum of SMN at each depth:

   Total SMN (kg ha^{-1}) = 0-30cm SMN (kg ha^{-1}) + 30-60cm SMN (kg ha^{-1}) + 60-90cm SMN (kg ha^{-1})

Worked example:

Sample depth	**Thickness of sampled layer (cm)**	**SMN conc. (mg kg^{-1})**	**Bulk density (kg m^{-3})**	**Weight of soil (t ha^{-1})**		**SMN (kg ha^{-1})**	
	Known depth	From analysis	Std value of 1.33kg m^{-3}	= 100 x thickness x bulk density		= SMN conc. x weight	
0-30cm	30	7.8	1.33	= 100 x 30 x 1.33	= 3990	= 7.8 x 3990	= 31.12
30-60cm	30	8.4	1.33	= 100 x 30 x 1.33	= 3990	= 8.4 x 3990	= 33.52
60-90cm	30	6.7	1.33	= 100 x 30 x 1.33	= 3990	= 6.7 x 3990	= 26.73
TOTAL							**91.37 kg N ha^{-1}**

Calculating over winter loss

While autumn SMN quantifies N at risk of loss, a simplified balance approach can be used to estimate over winter loss from autumn SMN, spring SMN and crop N. Assuming the system receives no inputs of N, loss from the soil (0-90cm) can be calculated using the following equation:

Over winter loss (kg N ha^{-1}) = Autumn SMN (kg N ha^{-1}) – Spring SMN (kg N ha^{-1}) – Crop N (kg N ha^{-1})

4.2.1.2 Porous Pot (PP) sampling

Sampling approach

Porous pots were installed prior to the first winter of sampling (harvest year 2007) in accordance with ADAS porous pot standard operating procedures derived from Lord and Shepherd, (1993) and Webster et al., (1993). Five pots were installed per field, located in a representative area of the field, away from atypical areas and headlands. Sampling was undertaken during the winters of 2006/7 and 2007/8 (harvest years 2007 and 2008), before and after the implementation of mitigation, commencing as the soil returned to field capacity just prior to the onset of winter drainage (as indicated by IRRIGUIDE (Bailey and Spackman, 1996)). Pots were sampled on a 2 weekly basis using a handheld vacuum pump to draw water into the pot. Samples were then refrigerated until analysis using an ion specific electrode. Sampling ceased when a sizeable soil moisture deficit developed, exposed by a lack of leachate following pressurisation. Where cultivation posed a risk to pots, tubing was buried alongside a magnet, and the location of pots recorded using sketch maps. Pots were located using a magnet detector and from measurements.

PP calculations

Analysis determined the nitrate concentration (mg N l^{-1}) of each sample. Results from each field on each sampling occasion were then averaged to produce a mean concentration. Leached loads were calculated by combining details of leachate concentration and drainage volume. In the absence of in situ drainage measurement (avoided due to its invasive determination), the field scale water balance model IRRIGUIDE (Bailey and Spackman, 1996) was used to estimate soil drainage using agro-meteorological, cropping, soil and cultivation details for each field.

The amount of drainage before the first sample, between all sampling occasions and after the last sampling date was calculated. Drainage between successive sampling dates was divided by two and one half apportioned to each of the two sampling dates (at the start and end of that drainage period). Drainage occurring before and after each sampling date was than summed to yield the total drainage for that date.

The total drainage associated with each sampling occasion was then multiplied by the corresponding average N concentration. Losses from each date were than summed to obtain the total N loss for that drainage season.

Worked example:

<table>
<tr><th>Sampling occasion</th><th colspan="2">1</th><th colspan="2">2</th><th colspan="2">3</th><th colspan="2">4</th><th colspan="2">5</th><th colspan="2">6</th><th colspan="2">7</th><th colspan="2">8</th></tr>
<tr><td>Ave. conc. (mg nitrate N l^{-1})</td><td colspan="2">7.6</td><td colspan="2">74.9</td><td colspan="2">52.2</td><td colspan="2">37.6</td><td colspan="2">21.2</td><td colspan="2">17.1</td><td colspan="2">9.6</td><td colspan="2">5.4</td></tr>
<tr><td>Drainage (mm)</td><td>9.2</td><td colspan="2">37.4</td><td colspan="2">68.8</td><td colspan="2">11.1</td><td colspan="2">97.6</td><td colspan="2">28.6</td><td colspan="2">2.8</td><td colspan="2">36.6</td><td>5.1</td></tr>
<tr><td>Split drainage (mm)</td><td>9.2</td><td>18.7</td><td>18.7</td><td>34.4</td><td>34.4</td><td>5.6</td><td>5.6</td><td>48.8</td><td>48.8</td><td>14.3</td><td>14.3</td><td>1.4</td><td>1.4</td><td>18.3</td><td>18.3</td><td>5.1</td></tr>
<tr><td>Apportioned drainage (mm)</td><td colspan="2">27.9</td><td colspan="2">53.1</td><td colspan="2">40</td><td colspan="2">54.4</td><td colspan="2">63.1</td><td colspan="2">15.7</td><td colspan="2">19.7</td><td colspan="2">23.4</td></tr>
<tr><td>Leached load (kg N ha^{-1})</td><td colspan="2">2.1</td><td colspan="2">39.8</td><td colspan="2">20.9</td><td colspan="2">20.5</td><td colspan="2">13.4</td><td colspan="2">2.7</td><td colspan="2">1.9</td><td colspan="2">1.3</td></tr>
<tr><td colspan="13">TOTAL LEACHED LOAD</td><td colspan="4">102.6 kg N ha^{-1}</td></tr>
</table>

4.2.1.3 SMN / PP Field selection

Limited resources meant SMN and PP sampling was restricted to a subset of MSA / EMEL fields. Fields were selected on the basis of cropping / over winter state, farm representation (for the provision of fertiliser recommendations), presence of field scale mitigation (namely cover crops and the delayed application of manure) and manure management (both past and present). The subset aimed to be broadly representative of each catchment, whilst including fields considered high risk, for example those previously in long term grass, rotational set-a-side, legumes, used by pigs or poultry, or receiving large amounts of manure. To maximise coverage, fields were initially restricted to SMN or PP sampling and not both. However in 2008 additional funding facilitated an extension of the sampling network; efforts were then made to match SMN and PP field selections to allow investigation into SMN – leached loss relationships. Accordingly 2008 subsets were larger than those in 2007. Details of the SMN and PP datasets, including those of '2007 fields only' are presented in Table 4-2 and Table 4-3.

Table 4-2: Crop type distribution of SMN fields – 'all fields' shown in normal text, original 2007 fields in brackets.

Current crop	**Empool / Eagle Lodge**								**Milborne St Andrew / Dewlish**							
	2007				**2008**				**2007**				**2008**			
	No. fields		**% dataset total**		**No. fields**		**% dataset total**		**No. fields**		**% dataset total**		**No. fields**		**% dataset total**	
Grass	20	*(18)*	24	*(25)*	21	*(19)*	25	*(26)*	2	*(2)*	6	*(7)*	5	*(2)*	9	*(7)*
Kale	-	-	-	-	-	-	-	-	-	-	-	-	1	-	2	-
Linseed	2	-	2	-	-	-	-	-	-	-	-	-	-	-	-	-
Maize	7	*(7)*	8	*(10)*	3	*(3)*	4	*(4)*	3	*(3)*	10	*(11)*	9	*(3)*	16	*(11)*
Peas	1	*(1)*	1	*(1)*	-	-	-	-	-	-	-	-	-	-	-	-
Spring Barley	8	*(8)*	10	*(11)*	11	*(9)*	13	*(13)*	3	*(2)*	10	*(7)*	12	*(8)*	21	*(29)*
Spring beans	2	-	2	-	-		-	-	-		-	-	-		-	-
Stubble turnips	-	-	-	-	1	*(1)*	1	*(1)*	-	-	-	-	-	-	-	-
Winter Barley	7	*(7)*	8	*(10)*	7	*(5)*	8	*(7)*	1	*(1)*	3	*(4)*	3	*(2)*	5	*(7)*
Winter Oats	1	*(1)*	1	*(1)*	2	*(2)*	2	*(3)*	1	*(1)*	3	*(4)*	4	*(2)*	7	*(7)*
Winter Rye	-	-	-	-	-	-	-	-	-	-	-	-	1	*(1)*	2	*(4)*
Winter Wheat	23	*(18)*	27	*(25)*	29	*(24)*	35	*(33)*	19	*(17)*	61	*(61)*	18	*(9)*	32	*(32)*
WOSR	13	*(12)*	15	*(17)*	9	*(9)*	11	*(13)*	2	*(2)*	6	*(7)*	3	*(1)*	5	*(4)*
Total	**84**	**(72)**			**83**	**(72)**			**31**	**(28)**			**28**	**(28)**		

Table 4-3: Crop type distribution of porous pot fields – all fields shown in normal text, original 2007 fields in brackets.

Current crop	Eagle Lodge/ Empool								Milborne St Andrew							
	2007				2008				2007				2008			
	No. fields		% dataset total		No. fields		% dataset total		No. fields		% dataset total		No. fields		% dataset total	
Grass	4	*(4)*	24	*(24)*	10	*(5)*	33	*(29)*	3	*(3)*	18	*(18)*	3	*(2)*	11	*(12)*
Italian Ryegrass	-	-	-	-	-	-	-	-	-	-	-	-	1	*(1)*	4	*(6)*
Linseed	-	-	-	-	-	-	-	-	1	*(1)*	6	*(6)*	-	-	-	-
maize	1	*(1)*	6	*(6)*	2	*(1)*	7	*(6)*	4	*(4)*	24	*(24)*	5	*(1)*	19	*(6)*
Set-a-Side	-	-	-	-	-	-	-	-	1	*(1)*	6	*(6)*	-	-	-	-
Spring Barley	3	*(3)*	18	*(18)*	4	*(4)*	13	*(24)*	1	*(1)*	6	*(6)*	6	*(2)*	22	*(12)*
Winter Barley	1	*(1)*	6	*(6)*	4	*(1)*	13	*(6)*	-	-	-	-	-	-	-	-
Winter Oats	-	-	-	-	-	-	-	-	-	-	-	-	1	*(1)*	4	*(6)*
Winter Wheat	6	*(6)*	35	*(35)*	6	*(4)*	20	*(24)*	5	*(5)*	29	*(29)*	10	*(9)*	37	*(53)*
WOSR	2	*(2)*	12	*(12)*	4	*(2)*	13	*(12)*	2	*(2)*	12	*(12)*	1	*(1)*	4	*(6)*
Total	**17**	**(17)**			**30**	**(17)**			**17**	**(17)**			**27**	**(17)**		

4.2.2 Catchment scale

At the catchment scale boreholes, wells, springs and streams were sampled, enabling measurement of N concentration in water bodies. Groundwater sampling was supported by determination of groundwater levels.

Sampling approach

Spring, stream, well and boreholes were sampled between summer 2006 – summer 2008. Samples were taken on a monthly basis reflecting the delayed and buffered responses associated with groundwater dominated catchments. Sample intervals were also in line with similar monitoring conducted by the Environmental Agency and other groundwater catchment studies (e.g. Bowes et al., 2009; Howden et al., 2009). Water levels were measured on a fortnightly basis during the same two year period.

Springs and stream samples were collected using a plastic sample bottle, submerged in the middle (width and depth) of the channel. The bottle was purged three times before collection of a final sample volume. Boreholes and wells supplying farms / home directly were sampled from farmyard taps; water was allowed to run clear before the sample bottle filled. A portable pump was used where sources were not connected to the surface. Samples were refrigerated until analysis via an ion specific electrode. Groundwater level was measured by lowering a water level sensor probe until an audible sound was heard and LED illuminated; depth was then read from the integrated measuring tape. Dip measurements were corrected to the metres above ordnance datum (mAOD), measured at the top of each well / borehole.

Site selection

Ground / surface water sampling locations were chosen from a list of sampling points identified during catchment visits and liaison with farmers / local residents. Sites were selected on the basis of accessibility, the seasonality of flow / water level, condition / possible contamination, equipment required for sampling, and the nature of activities in close proximity; sampling points adjacent to manure heaps, and stream with unrestricted animal access were avoided. The locations of ground / surface water sampling points are shown in Figures 4-1 and 4-2, and a summary of the catchment sampling approach presented in Table 4-4.

Table 4-4: Details of catchment sampling approach

	EMEL				**MSA**				**TOTAL**
	Sample and dip	**Sample only**	**Dip only**	**Subtotal**	**Sample and dip**	**Sample only**	**Dip only**	**Subtotal**	
Borehole	24	7	39	70	8	12	11	31	**101**
Well	5	0	2	7	9	9	11	29	**36**
Spring	-	10	-	10	-	5	-	5	**15**
Stream	-	12	-	12	-	7	-	7	**19**
Totals	29	29	41	89	17	33	22	72	**171**

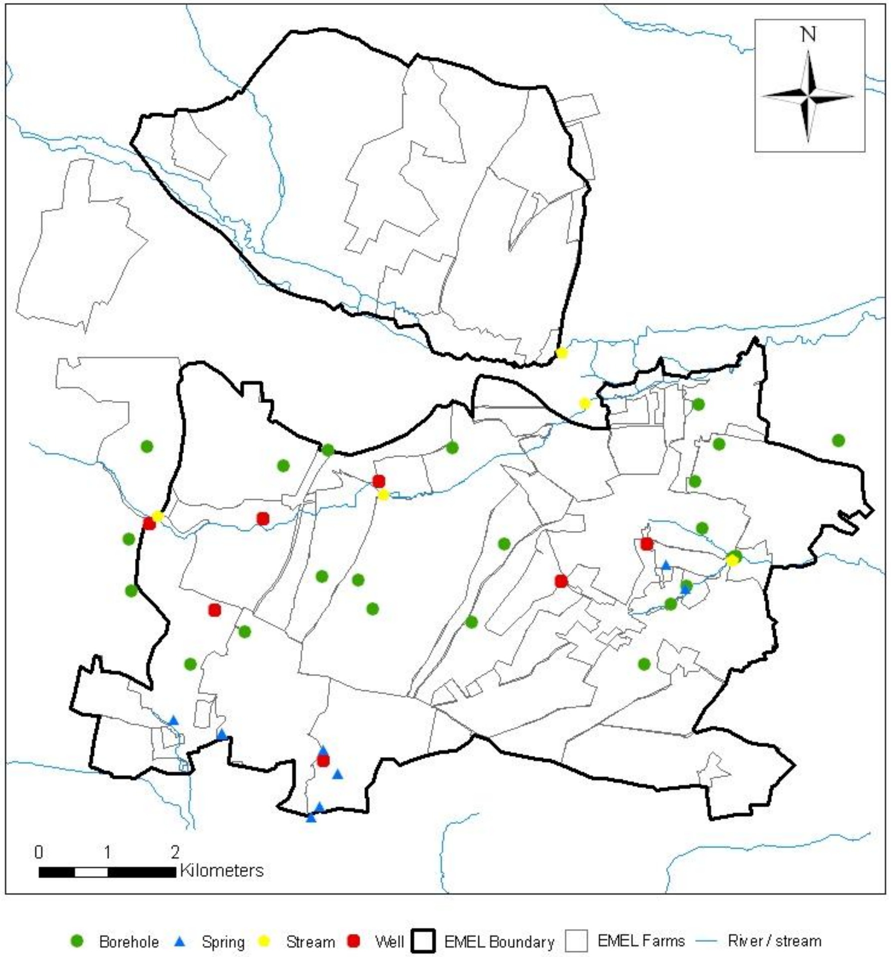


Figure 4-1: Location of EMEL sampling sites

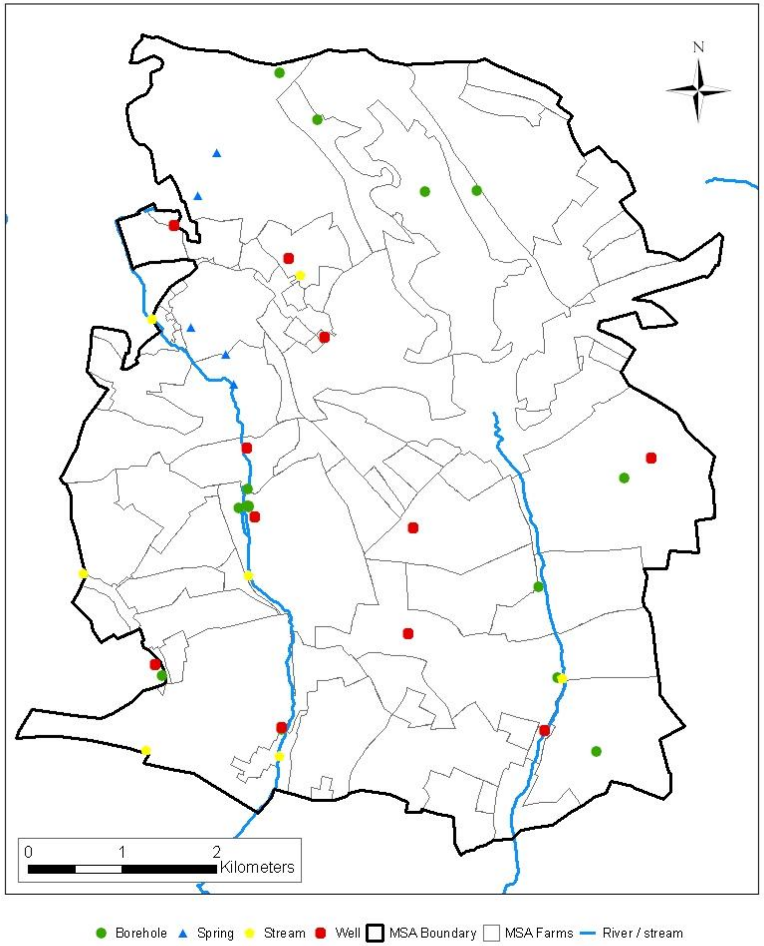


Figure 4-2: Location of MSA sampling sites

4.2.3 Data interpretation

4.2.3.1 Field scale

Fields were excluded from all analyses where details of the current / previous crop were absent. Results were also excluded from specific analyses where details of the factor being investigated were not available. Investigations focused on the eight most prevalent crops: grass, maize, spring malting barley (SBM), winter feed barley (WBF), winter oats (WO), winter feed wheat (WWF) and winter milling wheat (WWM) which covered c.90% of each catchment (see Figure 3-9). To ensure dataset consistency and facilitate valid comparisons, use of the 'all field' datasets was limited to analyses across both years combined and 2008 only. Where 2007 and 2008 results were compared, 'original fields' were employed to maintain a consistent dataset between years. In accordance with Lord et al., (1999) sensitivity to mitigation was investigated on both a 'before vs. after' basis in which pre- and post-mitigation results were compared, and on a 'with vs. without' basis using 2008 results only. The latter was useful where mitigation uptake was low and analyses benefitted from use of the larger 'all field' datasets.

4.2.3.2 Catchment scale

Catchment scale investigations focused on the temporal and spatial variability in water quality before and after mitigation. Due to inconsistencies in sampling dates between sites and catchments, water quality results were aligned to comparable sampling dates to produce comparable time series of N concentration in each catchment. Owing to more uniform sampling intervals in MSA, EMEL results were assigned to the closest MSA sample date. Where investigations were conducted on a site specific basis, complete datasets were utilised. Sites were excluded from analysis where more than 50% of comparable 'MSA sampling dates' remained empty following alignment; blanks resulted from large inconsistencies in sampling interval, un-sampled sites due to access problems, and possible sample contamination. Nitrate concentrations at MSA / EMEL sites were averaged on each 'aligned' sampling occasion to produce a mean time series of N concentration for each catchment. Time series were then compared before and after the adoption of mitigation (September – April). Assessments of mitigation impact were supported by calculations of mean and maximum concentrations across comparable sampling periods (September to May) and details of drinking water standard exceedances.

Spatial representations of results were used to illustrate the extent of spatial variability which may confound the usefulness of measurement in evaluating mitigation effectiveness when applied on a multi site basis.

4.2.3.3 Data analysis

Field scale results were subject to statistical analysis using GENSTAT v12. General ANOVAs were performed to test for significant differences in SMN and PP results before and after the implementation of mitigation. Autumn SMN, crop N, spring SMN, over winter loss, PP load and PP concentration were included as 'Y variates' whilst 'year' was the main treatment. Crop type and catchment were also included as treatments, and analysis conducted on an 'all interactions' basis to investigate crop / catchment specific responses to mitigation. To investigate responses to specific mitigation methods / management practices (e.g. manure applications), fields were classified according to the presence / absence of each practice, and the additional variant included as a factor in subsequent ANOVAs. For cover crops analysis was also conducted using 2008 data only to test for significant differences between fields with and without a cover crop. At the catchment scale descriptive statistics were calculated using Excel and ArcGIS V9.3 used to present results spatially.

4.3 Results

4.3.1 Field scale

4.3.1.1 Effect of mitigation on soil mineral nitrogen

Autumn SMN was significantly lower in 2008 than 2007 in both MSA and EMEL ($P<0.001$) (Figure 4-3). 'Previous crop' was close to explaining a significant amount of variation in both catchments ($p=0.055$ and 0.069 in MSA and EMEL respectively), however only in EMEL was a significant interaction between year and previous crop observed ($p<0.01$) with maximum reductions observed for grass and WOSR. Reductions in autumn SMN were generally larger in MSA than EMEL (for example on WBF fields autumn SMN decreased by 174.2 kg N ha^{-1} (78.5%) in MSA compared to 36.8 kg N ha^{-1} (31.6%) in EMEL); however responses were not significantly different between catchments.

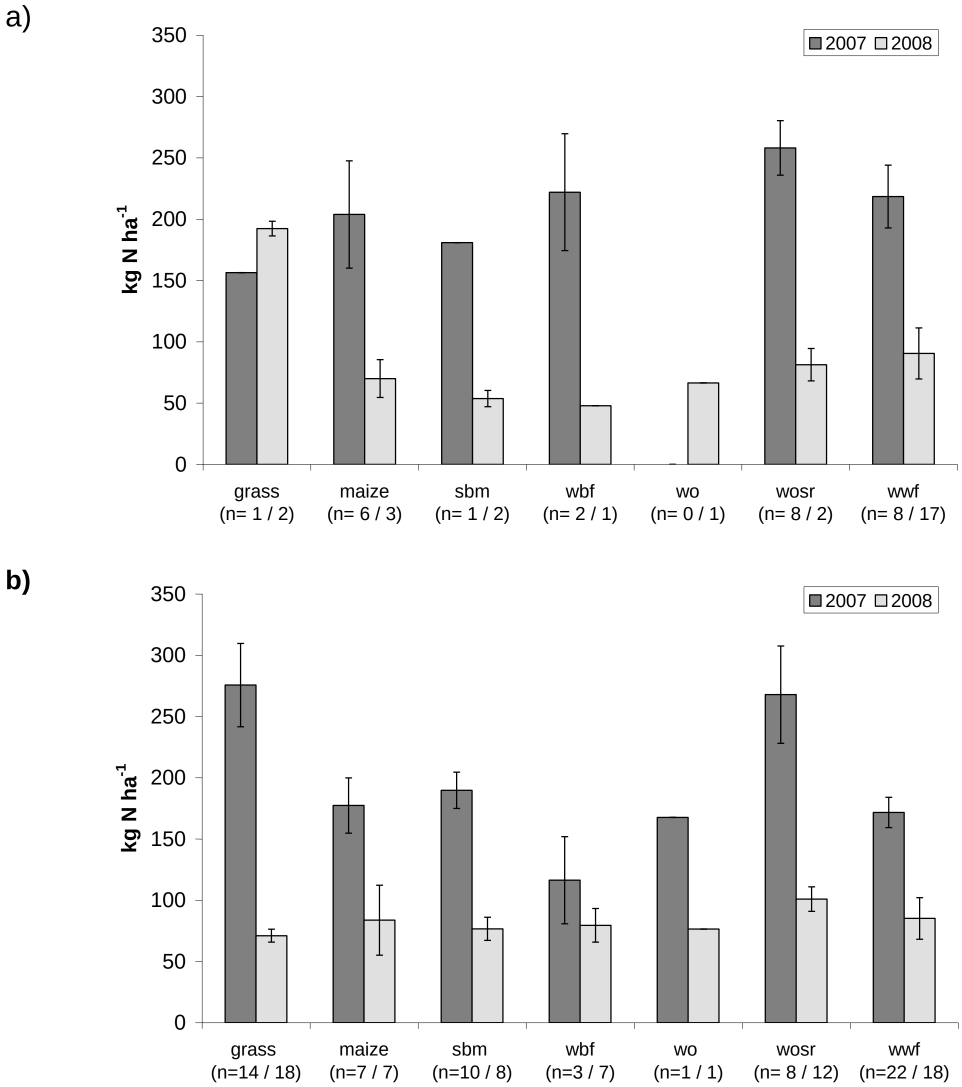
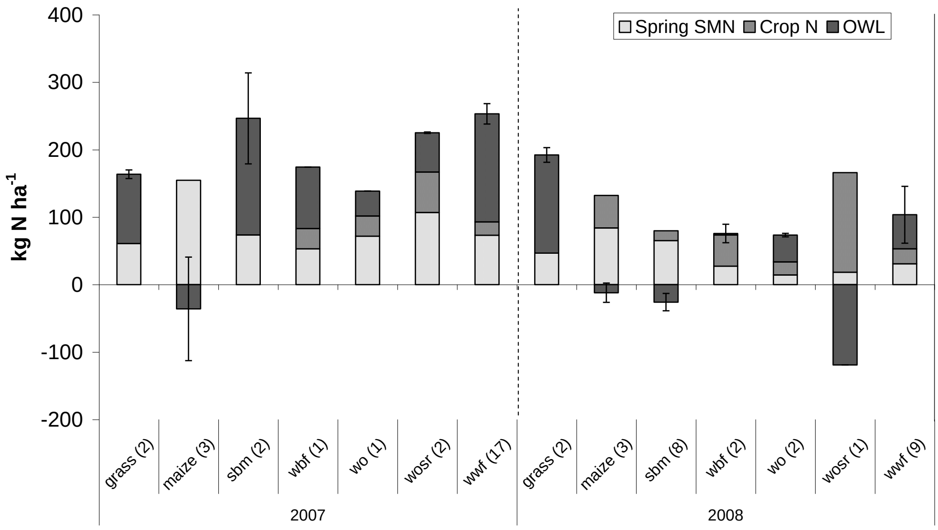


Figure 4-3: Autumn SMN (kg N ha^{-1}) before (2007) and after (2008) mitigation in a) MSA and b) EMEL. Mean and standard error shown on a previous crop basis with sample size in 2007 / 2008 shown in parentheses. Differences between years significant to p<0.001 in MSA and EMEL. Differences between crops not significant to p<0.05. Interactions between year and crop significant to p<0.01 in EMEL only. Differences between catchments and between catchment responses to mitigation not significant.

Considering the complete SMN balance (on a current crop basis) autumn SMN was significantly lower in 2008 than 2007 whilst crop N was significantly higher in 2008 (Figure 4-4). This in turn led to significantly lower over winter loss in 2008 in both catchments ($p<0.01$ / $p<0.001$ in MSA / EMEL) with maximum improvements associated with SBM. Significant interactions between year and crop type were limited to crop N ($p<0.001$ in MSA and EMEL) suggesting responses to mitigation were not crop type dependant. Significant differences in autumn SMN, crop N and over winter loss were however observed between current crop type in MSA. In EMEL current crop explained a significant amount of the variation in crop N only.

a)



b)

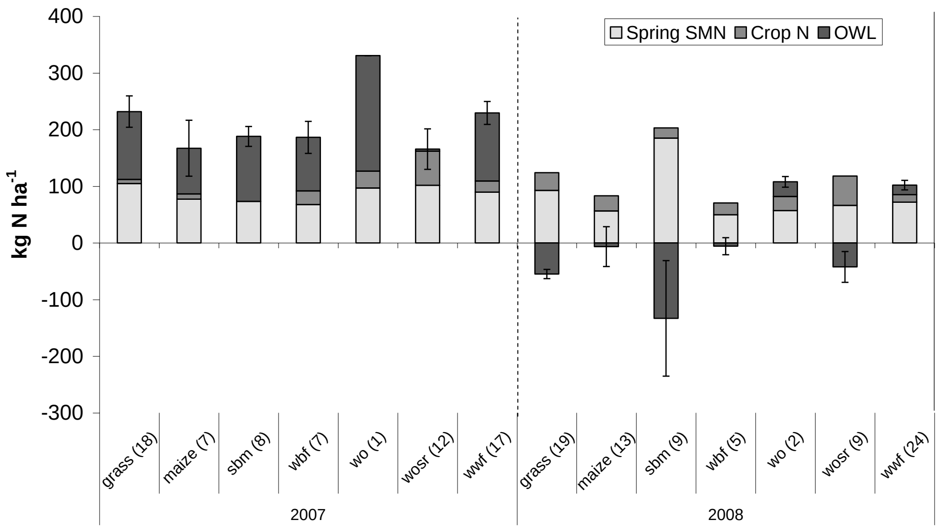


Figure 4-4: SMN balance (spring SMN, crop N and over winter loss (OWL) (kg N ha^{-1})) before (2007) and after (2008) mitigation in a) MSA and b) EMEL. Mean values presented on a current crop basis with sample size shown in parentheses. The total height of each bar equates to the autumn SMN (kg N ha^{-1}) except where OWL is negative, in which case autumn SMN is equal to the total positive bar height minus OWL. Error bars show standard error for over winter loss. Autumn SMN, spring SMN (EMEL only), crop N and OWL significantly different before and after mitigation in a) and b)($p< 0.01$ in all cases). In MSA autumn SMN, spring SMN crop N and OWL significantly different between crop types ($p<0.001$, except for spring SMN where $p<0.05$). In EMEL only crop N significantly different between crop types ($p<0.001$). Significant Crop – year interactions for crop N in both catchments ($p<0.001$). Significant differences between catchments, and interactions involving catchment observed for crop N only ($p<0.001$).

Effect of manure management

The application of manure had no significant effect on autumn SMN or over winter loss (MSA only due to lack of EMEL manure data) (Figure 4-5) Similarly, adoption of manure management plans (MMPs) had no significant impact on autumn SMN (after previous crop and year) or over winter loss (after current crop and year) from MSA fields. However it is worth noting that adoption of farm scale mitigation did not necessarily correspond with improved nutrient management in all fields and that a small dataset was associated with these results.

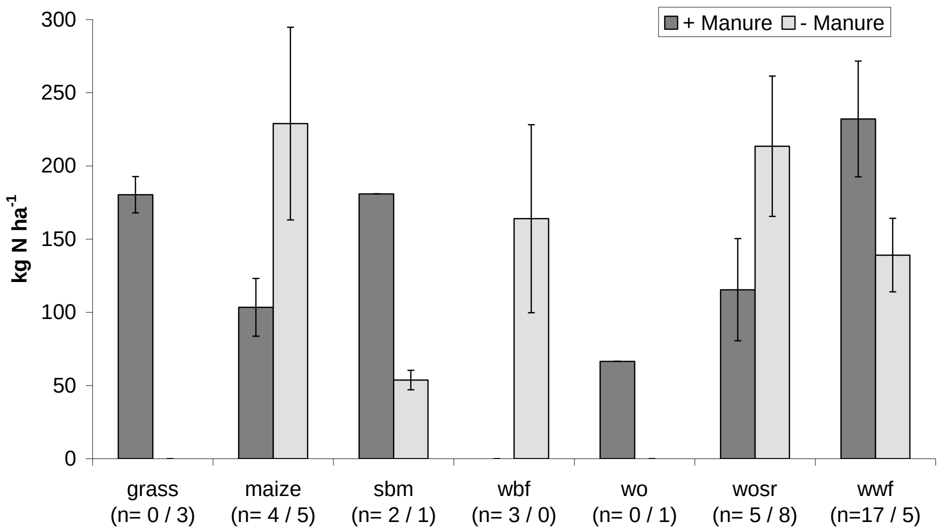


Figure 4-5: Effect of manure application on autumn SMN (kg N ha^{-1}) presented on a previous crop basis. Mean and standard error shown with sample size in parentheses. Manure application refers to applications made to previous crop during the preceding year. Differences + / - manure not significant. (MSA fields only due to missing manure data for some EMEL fields)

Effect of cover crops

The presence of cover crops in spring cropped fields tended to reduce soil N and loss. Across all spring cropped fields over winter loss was significantly lower ($p<0.001$) in fields where cover crops were grown compared to those without (Figure 4-6). This stemmed from significantly higher crop N ($p<0.001$) and significantly lower autumn SMN ($p<0.05$) in cover cropped fields. Differences in spring SMN in fields with / without a cover crop were not significantly different.

On a crop by crop basis, cover crops had a significant effect on SBM fields only. Over winter losses from maize was 34.6kg N ha^{-1} lower where a cover crop was grown, however large variation meant differences +/- cover crop were not significant (Figure 4-6). Crop N, averaging 31.7kg N ha^{-1} on maize fields with cover crops, accounted for the c. 31kg N ha^{-1} difference in spring SMN. The reduction in loss can therefore be attributed to a 34kg N ha^{-1} reduction in autumn SMN. For spring barley the 76.2 kg N ha^{-1} difference in over winter loss +/- cover crop remained significant when analysed independent of maize (p=0.014) (Figure 4-6). Cover cropped fields retained 43.7kg N ha^{-1} in plant matter, and recorded 8.5kg N ha^{-1} higher spring SMN. In contrast to maize, cover crops in SBM fields did not directly offset differences in spring SMN + / - a cover crop. Autumn SMN was again lower where cover crops were established, however differences +/- cover crops were smaller on SBM than maize fields.

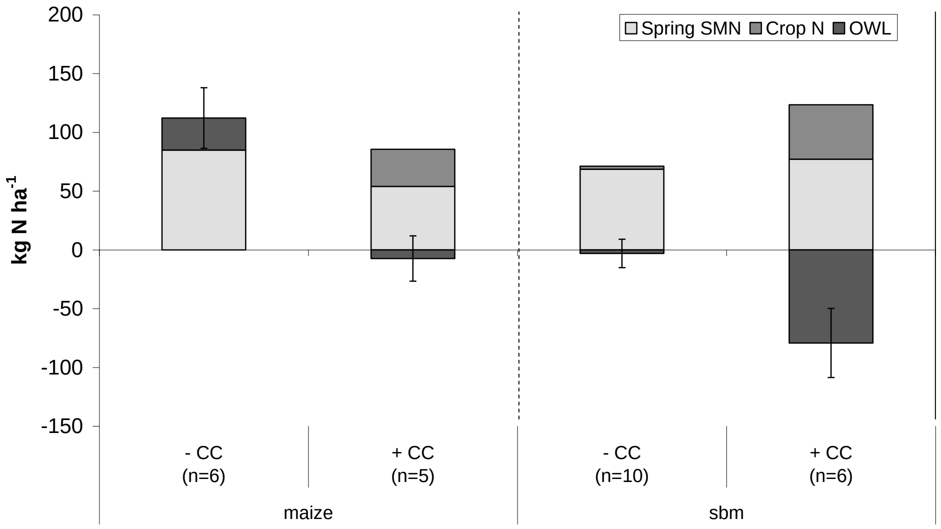


Figure 4-6: Effect of cover crops on the SMN balance (spring SMN, crop N and Over Winter Loss (OWL) (kg N ha^{-1})) for maize and SBM fields in 2008. The total height of each bar equates to the autumn SMN (kg N ha^{-1}) except where OWL is negative – see Figure 4-4. for details. Figure shows mean values on a current crop basis with sample size shown in parentheses; error bars relate to OWL only. Differences in autumn SMN, crop N and over winter loss with (+) / without (-) cover crops (cc) significant to p<0.05. Differences in OWL between crops, and crop x +/- cc interactions not significant. If analysed on an individual crop basis differences in over winter loss with / without cover crops (cc) significant for SBM only.

Over winter loss was significantly lower from cover cropped fields than other over winter states (grass, stubble and winter crops) (Figure 4-7). Largest over winter losses were associated with winter crops due to significantly higher autumn SMN and lower spring SMN; differences between autumn and spring SMN were too great to be offset by moderate crop N. Grass and stubble returned intermediate ‘losses’ of +35.6 and +8.6kg N ha^{-1} respectively. However, with autumn SMN closely related to the previous crop, comparisons should also be made irrespective of autumn SMN. Assuming equal autumn SMN, over winter loss from cover cropped fields was c.10 kg N ha^{-1} lower than from grass fields, and around 45 kg N ha^{-1} lower than from stubble and winter cropped fields. With cover cropped fields occupying spring cropped fields which would otherwise be in stubble, comparisons with the latter are most valid.

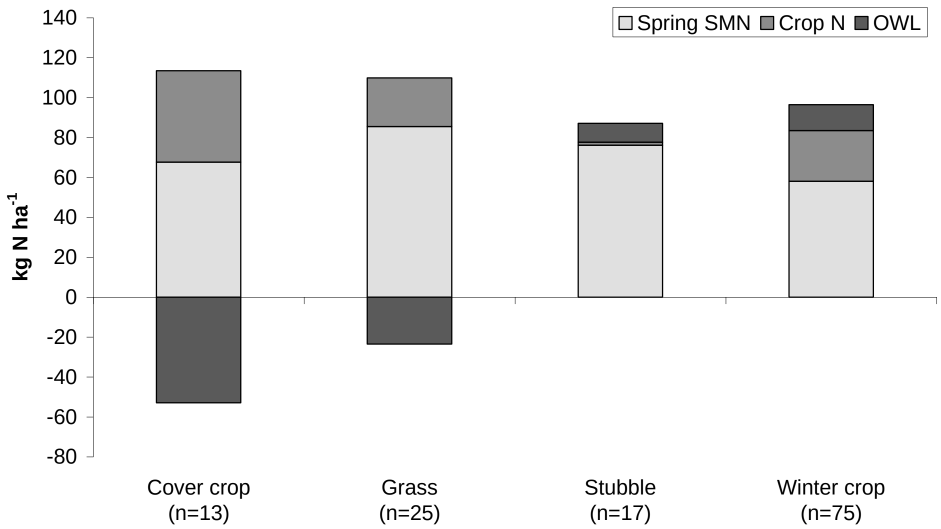


Figure 4-7: Effect of over winter state on SMN balances (spring SMN, crop N, Over Winter Loss (OWL) (kg N ha^{-1})). Mean and standard error shown with sample size in parentheses. The total height of each bar equates to the autumn SMN (kg N ha^{-1}) except where OWL is negative – see Figure 4-4 for details. Differences in autumn SMN, spring SMN and crop N between over winter states significant to $p<0.05$.

4.3.1.2 Effect of mitigation on porous pots

PP concentration and leached load were significantly different before and after the implementation of mitigation ($p<0.05$). However in marked contrast to the large reduction in SMN shown in Figure 4-3, loads and concentrations tended to be higher in 2008 than 2007 (Figure 4-8). The number of fields with mean concentrations in excess of the DWS was 29% higher in 2008 than 2007 (68% and 88% respectively) across MSA and EMEL combined.

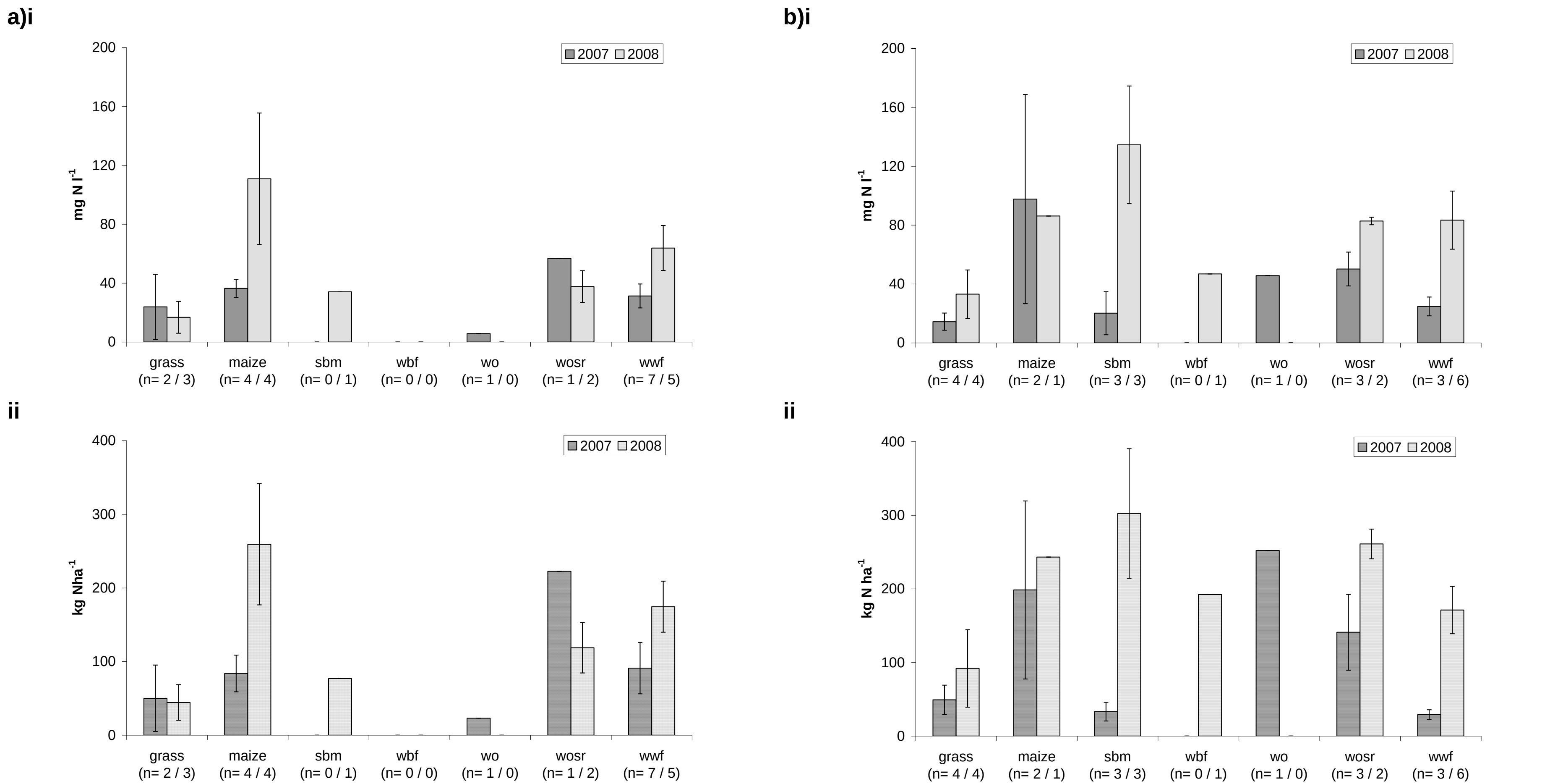
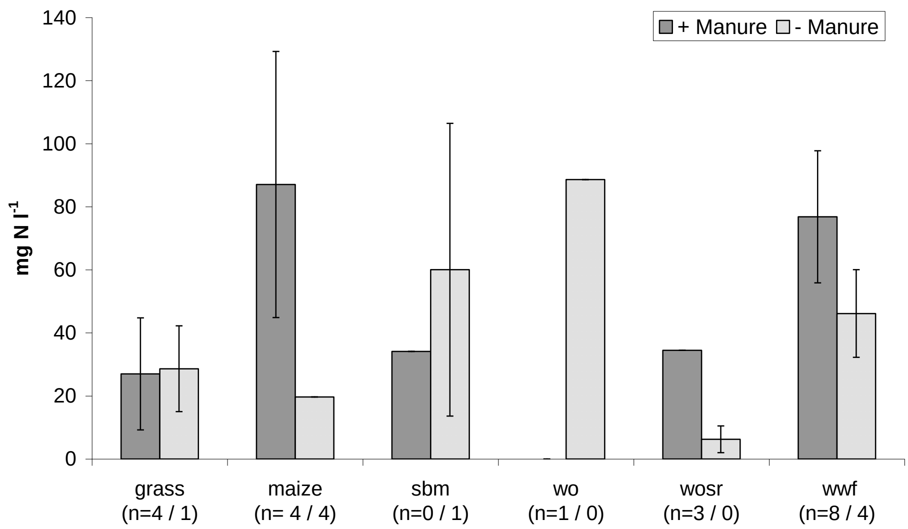


Figure 4-8: PP concentration (mg N l^{-1}) (i) and PP leached load (kg N ha^{-1}) (ii) in a) MSA and b) EMEL before (2007) and after (2008) mitigation. Mean and standard error presented on a previous crop basis with sample sizes in 2007 / 2008 shown in parentheses. Differences between years significant to $p<0.05$ in MSA and $p<0.001$ in EMEL. Differences between crop type significant to $p<0.05$. No significant interactions between year and crop type, and no significant differences or interactions involving catchment.

Crop type displayed no significant interaction with year however significantly larger concentration / loads were associated with maize and WOSR, and lowest loss / concentration with grass. Maize resulted in especially high concentrations, however leached load results were less extreme. Responses to mitigation were not catchment specific with no significant interaction observed between year and catchment. However evidence of an overall increase in loss / concentration was less evident on a catchment specific basis especially in MSA where concentration and load decreased on grass and WOSR fields. In EMEL improvements were restricted to maize (concentration only).

a)



b)

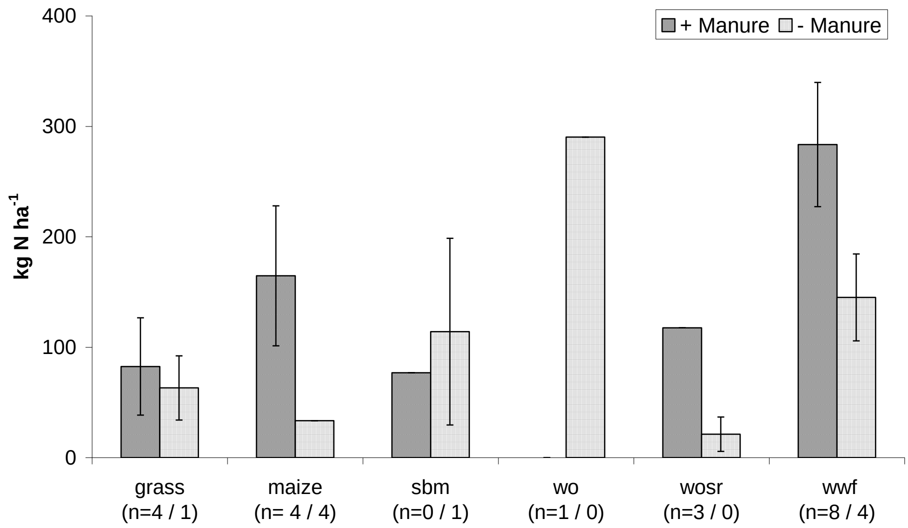


Figure 4-9: Crop average a) concentration (mg N l^{-1}) and b) leached load (kg N ha^{-1}) in fields + / - manure. Mean and standard error presented on a current crop basis with sample size in 2007 / 2008 shown in parentheses. Manure application refers to applications made to the current crop. Differences + / - manure not significant to $p<0.05$.

Effect of manure management

As was the case for SMN results, data availability restricted investigations into the effect of manure on N loss to MSA fields only. Applications of manure within the same year were found to have no significant effect on mean concentration or leached load (Figure 4-9 – results on a previous crop basis not shown) with inconsistent responses between crops, and concentration / load results not fully in agreement. Only for leached load, analysed on a current crop basis, were differences close to being significant (p=0.056), and losses generally higher where manure was applied. Assessments of MMP impact were limited to WWF. Smaller increases in losses corresponded with the adoption of MMPs but differences were not significant.

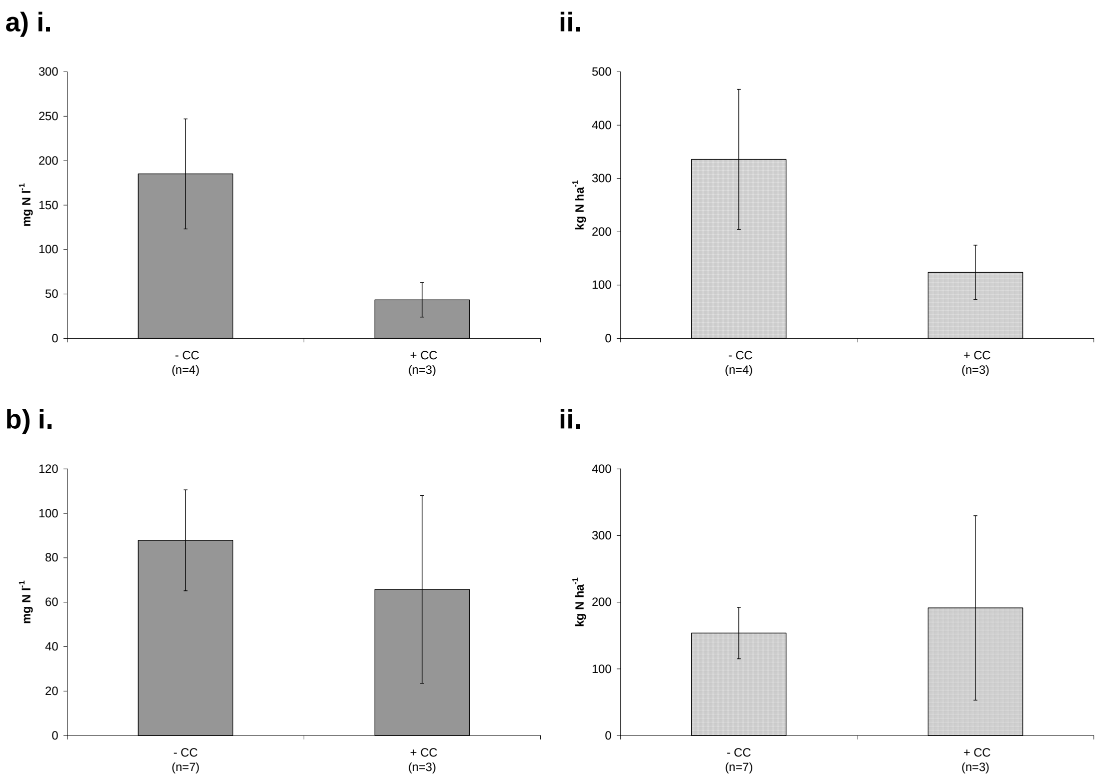


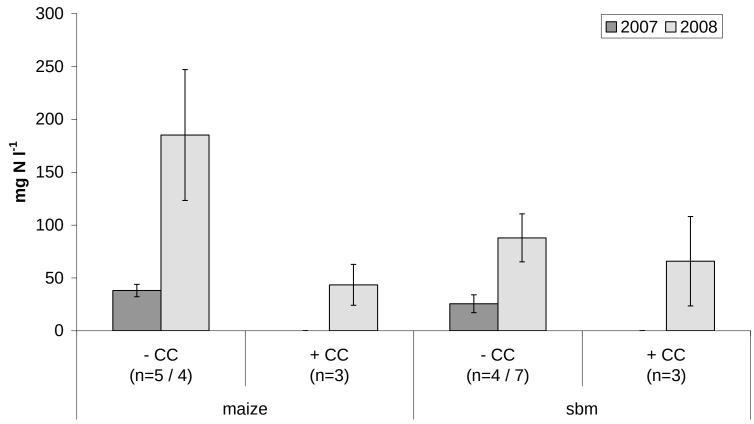
Figure 4-10: Comparison of i) concentration (mg N l^{-1}) and ii) leached load (kg N ha^{-1}) from a) maize and b) SBM fields + / - cover crops (cc) in 2008. Mean and standard error shown with sample size in parentheses. Differences + / - cover crop not significant in all cases.

Effect of cover crops

Cover crops grown before spring crops had a positive but insignificant effect on concentration and leached load (Figure 4-10). Average concentrations were

141.7mg N l^{-1} and 22.1mg N l^{-1} lower where cover crops preceded maize and SBM respectively, however wide variability, especially for SBM, meant differences were not significant to $p<0.05$. In terms of leached load, a positive but again insignificant response was observed for maize. Although results were not significant it is interesting to note that in contrast to SMN results cover crops had greater impact on losses from maize than SBM (see Figure 4-6). Analysing results on a before vs. after basis, losses and concentration were significantly lower pre-mitigation even where a cover crop was grown (Figure 4-11). Losses and concentration from maize cover cropped fields were however close to those observed pre-mitigation. Given the large increase in losses / concentration between pre- and post-mitigation years for non cover cropped fields, this likeness highlights the potential for sizeable reductions in concentration / load associated where cover crops are grown.

a)



b)

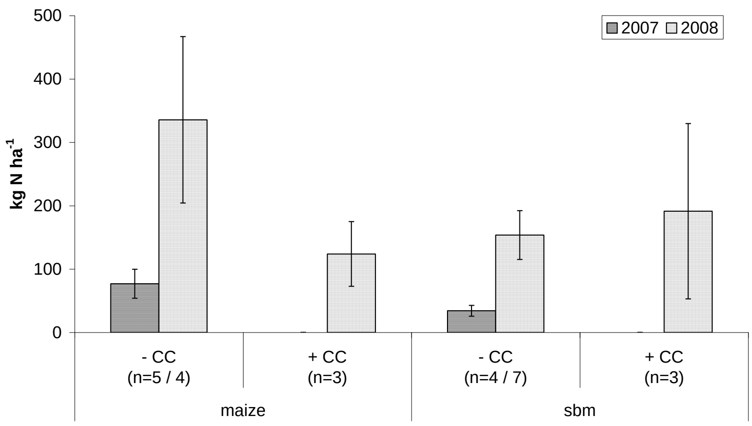
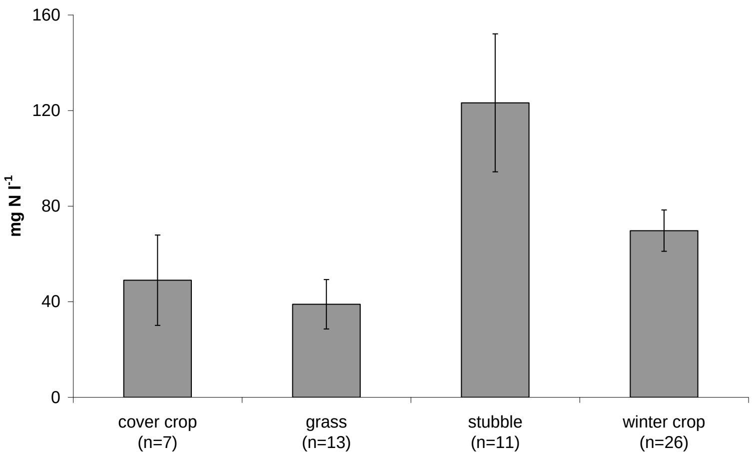


Figure 4-11: a) Concentration (mg N l^{-1}) and b) leached load (kg N ha^{-1}) from spring cropped fields + / - cover crops (cc). Mean and standard error shown with 2007 / 2008 sample sizes in parentheses. Differences between years significant to p<0.05. Differences between crops and + / - cc not significant. Interactions between crop, year and presence of cover crop not significant.

Comparison of losses from cover cropped fields with other over winter states (grass / winter crops / stubble) highlighted the effectiveness of grass cover in reducing losses (Figure 4-12). Concentration and load were significantly lower from grass than cover crops, a finding which differs from that observed in SMN over winter loss (see Figure 4-7). However comparisons between cover crop and stubble fields are perhaps more meaningful given that both are associated with spring crops. The establishment of grass represents a change in cropping rather than a modification to an existing rotation. Leached loads and concentrations were significantly lower from cover crops than stubble fields.

a)



b)

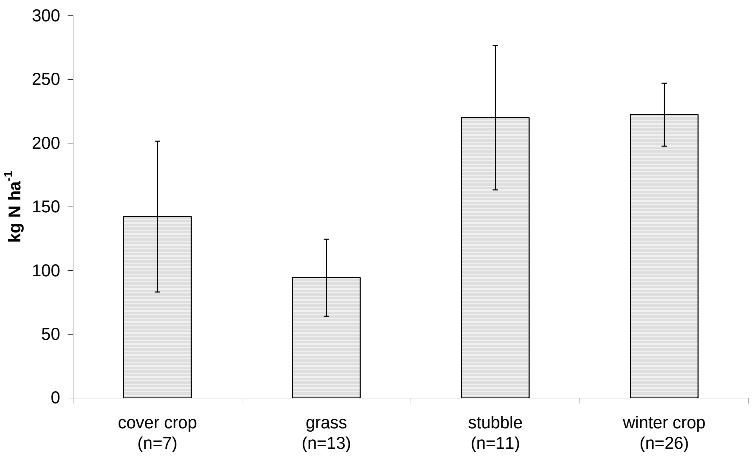


Figure 4-12: Comparison of a) concentrations (mg N l^{-1}) and b) leached loads (kg N ha^{-1}) between over winter states in 2008. Mean and standard error shown with sample size in parentheses. Differences between over winter states significant to $p<0.001$.

4.3.2 Effect of mitigation on catchment scale measurements

Nitrate concentration averaged across all sites and sampling occasions was 2.1mg N l^{-1} (32%) higher in EMEL than MSA. The number of EMEL samples exceeding the drinking water standard was more than double that in MSA (19.5 vs. 9.4%), and the dataset maxima 14.2mg N l^{-1} higher in EMEL than MSA. Following the implementation of mitigation in 2008, average N concentrations over comparable sampling periods decreased by 0.8mg N l^{-1} (8%) in EMEL but increased in MSA by 0.1mg N l^{-1} (2%) (Table 4-5). Accordingly the number of samples exceeding the drinking water standard fell in EMEL only. However a reduction in the maximum concentration was observed in MSA only.

Table 4-5: N concentrations (mg N l^{-1}) before (2007) and after (2008) the implementation of mitigation. Results averaged across all sites and sampling occasions. % exceedance refers to the number of individual samples exceeding the drinking water standard of 11.3mg Nl^{-1}.

	MSA		EMEL	
	Before	**After**	**Before**	**After**
Mean	6.77	6.89	9.24	8.46
Max	34.21	31.65	41.90	45.60
Min	0.20	0.30	0.20	0.20
Range	34.01	31.34	41.70	45.40
SE	0.35	0.29	0.32	0.26
% exceedance	9.2	11.6	21.6	19.3

Time series of N concentration (averaged across all sites) revealed no clear response to mitigation in either catchment (Figure 4-13) and although concentrations were generally lower post mitigation in EMEL (across comparable sample periods of September to May) the opposite was true for MSA. Variability in N concentration time series could in the most part be attributed to variation in rainfall with maximum concentrations corresponding with high rainfall, typically during the winter months. High rainfall was also observed during summer 2007 which received 188mm (79%) more rain than during the same period (May – Sept) in 2006. As a result N concentrations at MSA sites were higher during summer 2007 than summer 2006. In EMEL peak summer rainfall corresponded with a distinct peak in N concentration. During comparable sampling periods rainfall was 121mm (17%) lower after mitigation.

On a site by site basis mean concentrations (across comparable sampling periods of September to end April) decreased at 44% and 72% of sites, and maximum concentrations at 44 and 62% of sites in MSA and EMEL respectively. Improvements in mean concentration averaged 0.9mg N l^{-1} (13%) in MSA and 1.0mg N l^{-1} (8%) in EMEL (averaged across 'improved' sites only). Improvements were however too modest for many sites to achieve lower concentration 'categories' post mitigation as shown in Figures 4-14 to 4-17. Figure 4-14 - Figure 4-17 do however demonstrate the degree of spatial variability in nitrate concentrations especially in MSA.

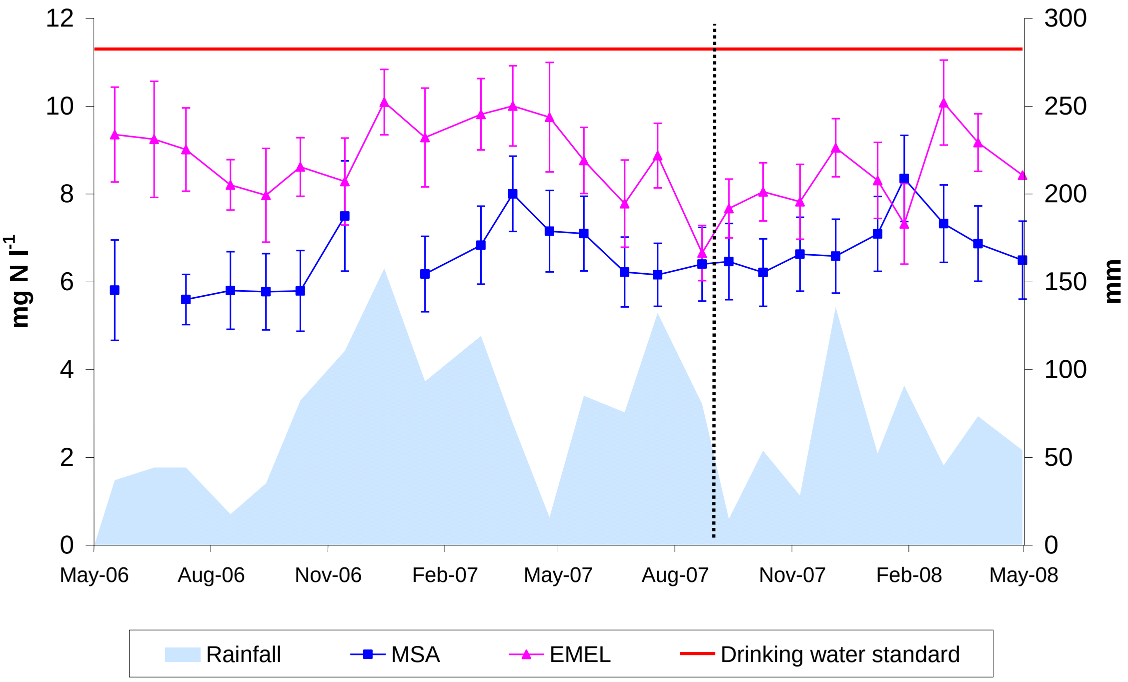


Figure 4-13: Temporal variability in groundwater N concentration (mg N l^{-1}) in MSA and EMEL. Dashed line marks the implementation of mitigation. Rainfall (mm) and the 11.3 mg N l^{-1} drinking water standard are also shown.

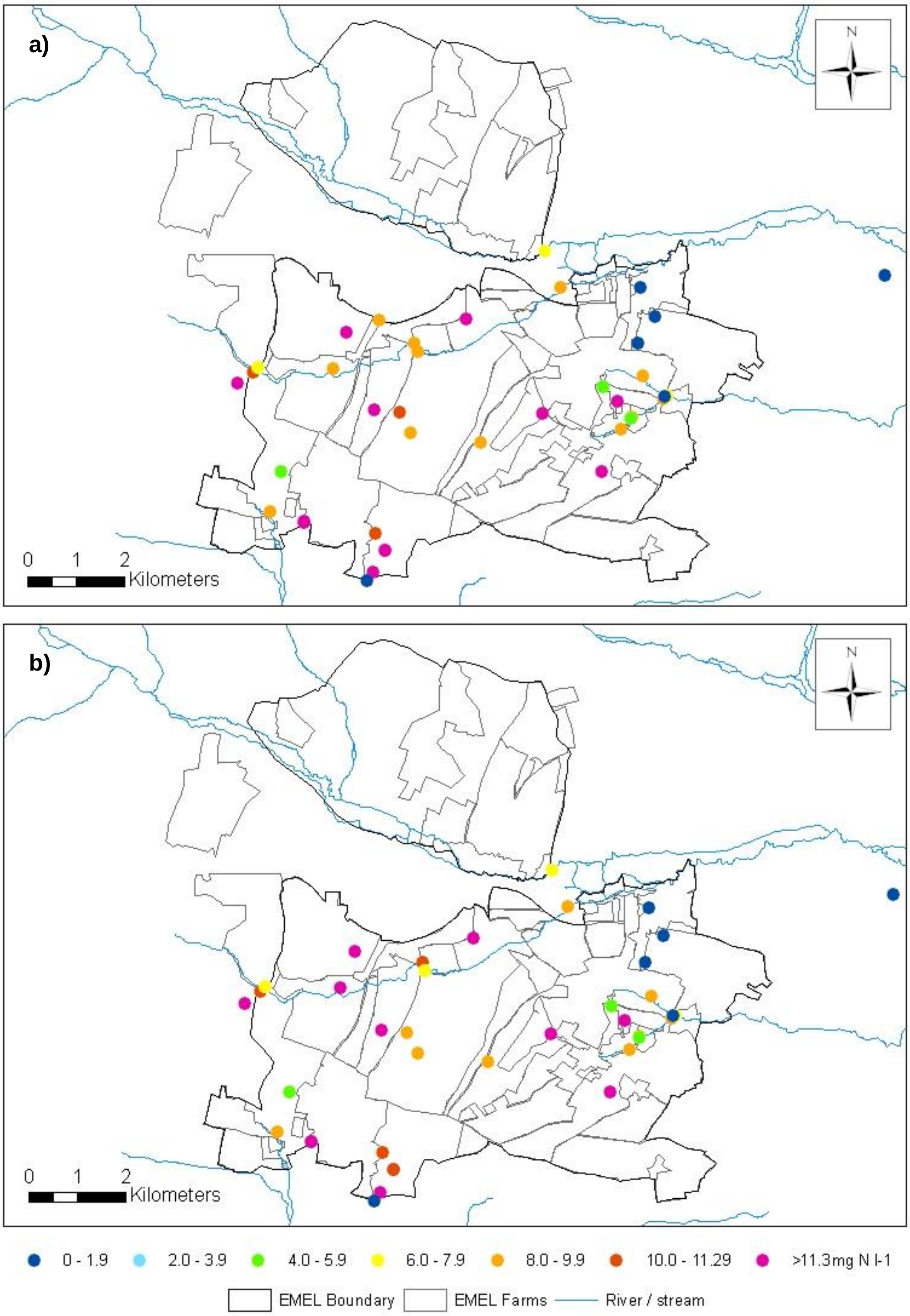


Figure 4-14: Average N concentration (mg N l^{-1}) at sampling sites in EMEL a) before and b) after mitigation (between comparable sampling periods of September to end April)

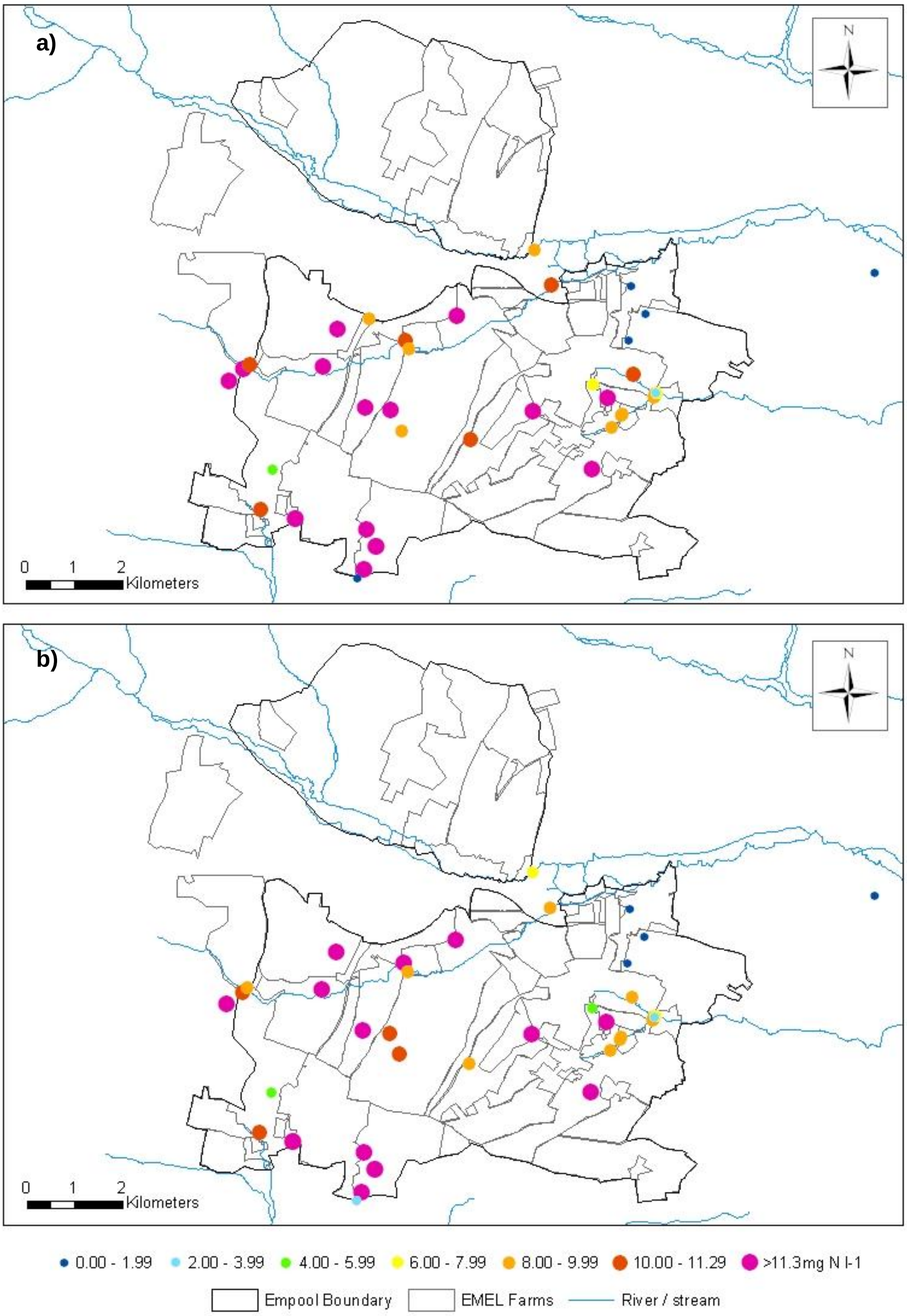


Figure 4-15: Maximum N concentrations (mg N l^{-1}) a) before and b) after the implementation of mitigation in EMEL (between comparable sampling periods of September to end April)

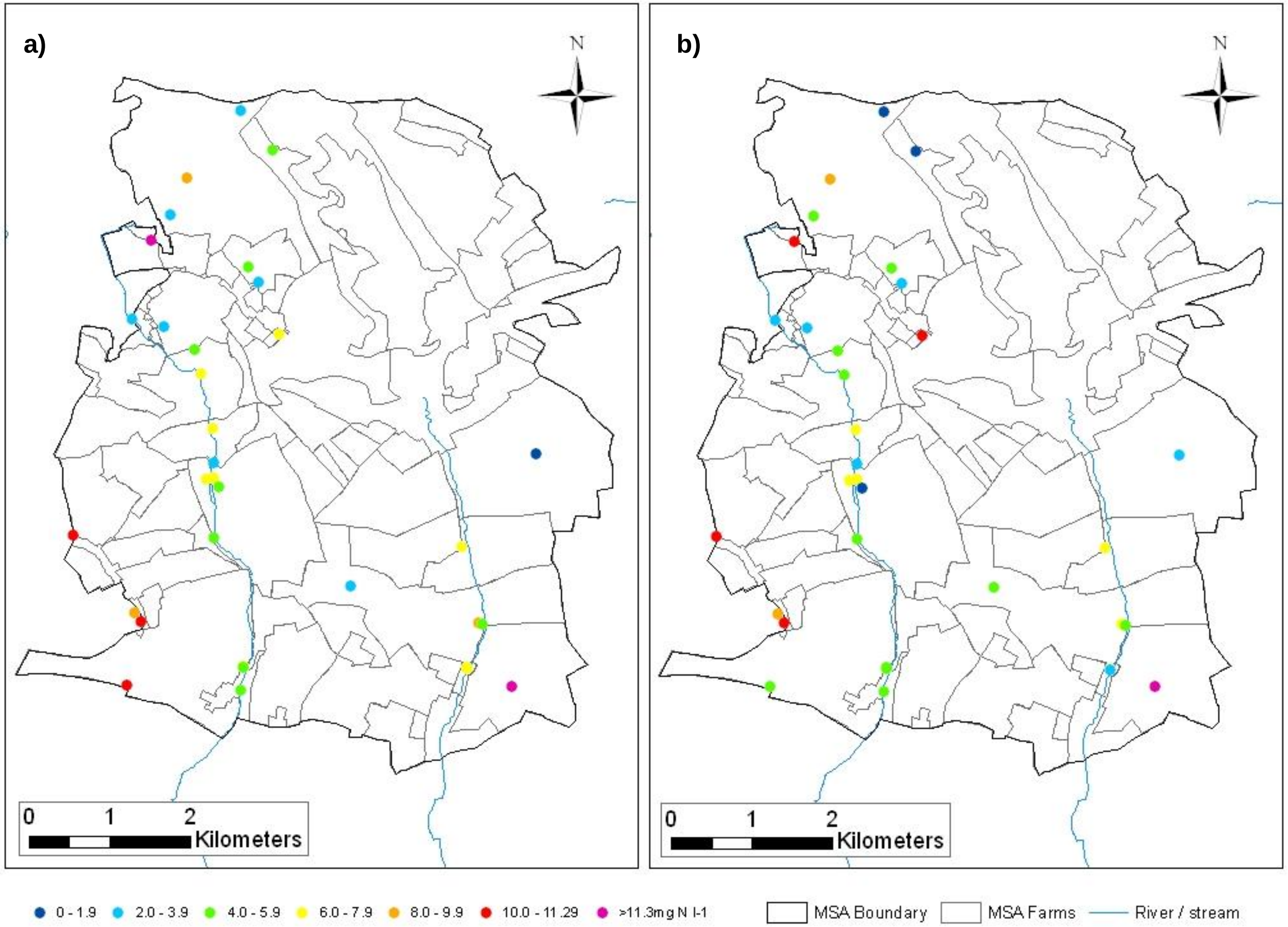


Figure 4-16: Average N concentration (mg N l^{-1}) at sampling sites in MSA a) before and b) after mitigation (between comparable sampling periods of September to end April)

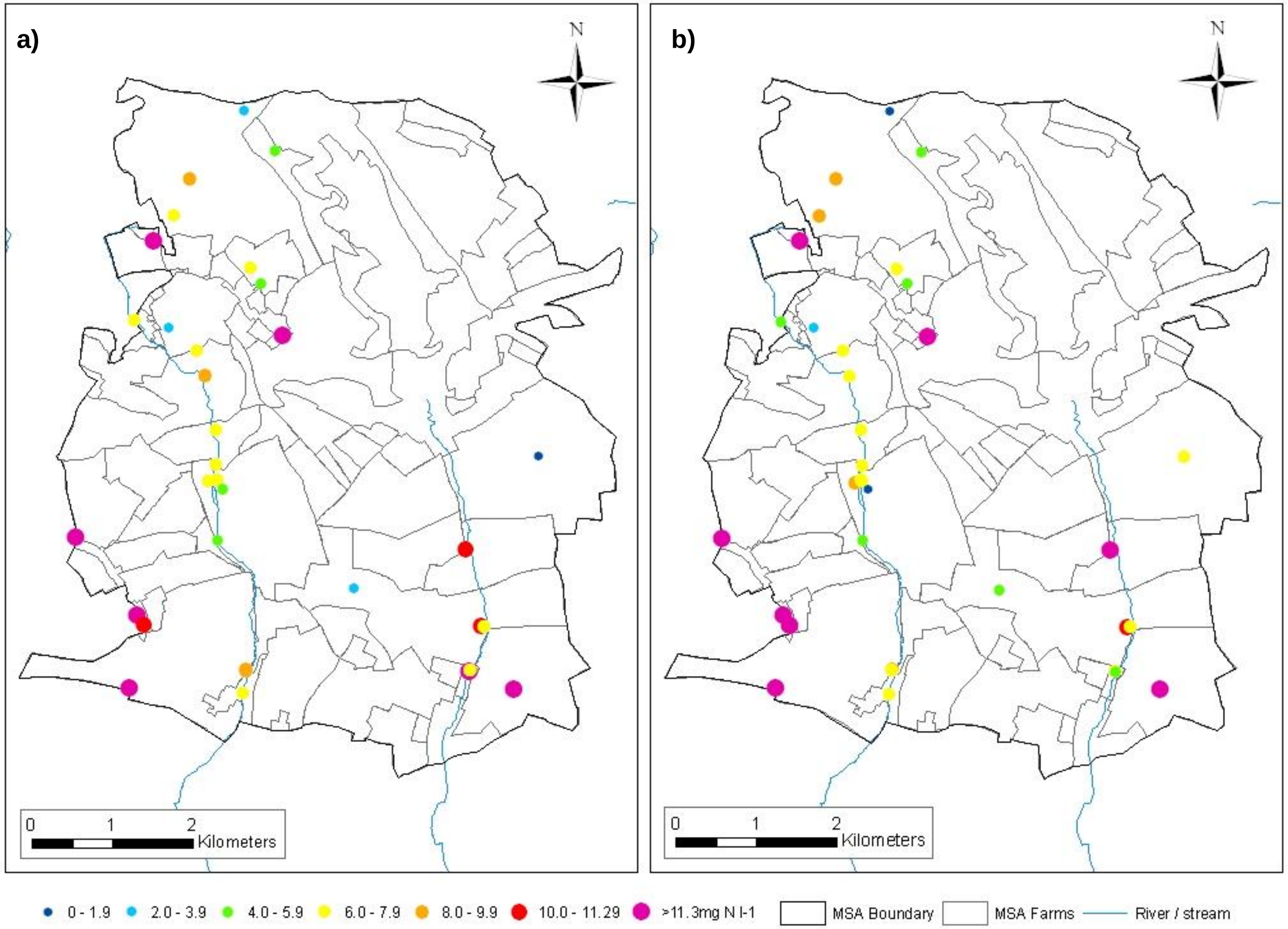


Figure 4-17: Maximum N concentration (mg N l-1) at sampling sites in MSA a) before and b) after mitigation (between comparable sampling periods of September to end April)

4.4 Discussion

4.4.1 Sensitivity of measurement to mitigation

Measurement at the field and catchment scale exposed no consistent response to mitigation. SMN and PP results were both significantly different following the implementation of mitigation however only SMN results displayed a positive response. At the catchment scale a reduction in mean concentration and total DWS exceedances was observed in EMEL but not MSA, and improvements on a site by site basis were modest. The lack of consistency in results suggests an overriding sensitivity to other factors besides mitigation, namely environmental variability, and minimal mitigation induced change; the short timescales over which assessments were made will have exacerbated the implications of these factors. Measurement approaches appeared more successful where individual mitigation methods were analysed on a 'with vs. without' basis (e.g. + / - cover crops) and the issue of annual variability removed. However with this comes a need to account for spatial variability. Due to scale and methodological differences, further discussions are provided on a method by method basis below.

4.4.1.1 Field scale measurement

Substantial reductions in autumn SMN initially suggested a positive response to mitigation, however the modest degree of mitigation induced change and contradictory PP suggests other factors besides mitigation are more likely to explain the observed improvements. Shepherd (1999) attributed differences in autumn SMN between years to differences in the N balance (inputs – offtake); however, yields were generally higher in 2006 than 2007 conducive to lower SMN (through increased crop uptake) in 2007 (measured autumn / winter 2006). Marked differences in rainfall between years offer a more likely explanation of observed results. This is in agreement with Beckwith et al., (1998) and Johnson et al., (2002) who attributed differences in PP concentration to annual variability in rainfall.

Rainfall during the 2007 summer (May – September) was 79% higher than in 2006, facilitating high summer leaching and resulting in the low autumn SMN observed in 2008 (sampled late autumn 2007). While low SMN would suggest low leached losses, it can be postulated that a dry, warm autumn in 2007 allowed mineral N pools to replenish prior to the late onset of winter drainage; accordingly peak concentrations were observed in December / January. Furthermore, over winter rainfall (November – March) was slightly lower in 2007/8 which in accordance with Beckwith et al., (1998), Shepherd (1999) and Johnson et al., (2002) induces higher concentrations due to incomplete leaching and reduced dilution. It should also be noted that PP samples from early winter sampling occasions were lost in 2006 (2007 results), and with highest concentrations observed early in the drainage season (Goulding et al., 2000), peak concentrations are likely to have been missed. Sensitivity to rainfall and other environmental factors that resulted in contrasting SMN and PP responses demonstrates the difficulty in using measurement for short term evaluations.

SMN and PP results were more consistent when used to analyse field scale mitigation on a 'with vs. without' basis in which annual variability can be avoided. Positive responses to cover crops were observed in SMN over winter loss, PP load and PP concentration. In agreement with Beaudoin et al., (2005) autumn SMN was lower where a cover crop was grown, a result of reduced mineralisation and increased nutrient uptake. Crop N was significantly lower in the presence of a cover crop, averaging 37.2kg N ha^{-1}; this is within the range reported by Beaudoin et al., (2005) and close to the 39kg N ha^{-1} observed by Vos and van der Putten (2004). As a result over winter loss was significantly lower where cover crops preceded spring crops. PP concentrations were 63% and 25% lower where cover crops were sown before maize and SBM respectively, in agreement with Lord et al., (1999) and others (see Table 4-6). However differences in PP results + / - a cover crop were not significant and reductions in leached load were limited to maize only. SMN and PP results also highlighted the benefit of grass in reducing over winter loss, as reported by Catt et al., (1998), Shepherd and Webb (1999) and Lord et al., (2007). Continual crop cover and nutrient uptake under grass resulted in significantly lower over winter loss, leached loads and concentration than observed from winter crops / stubble. The relative benefit of cover crops and grass differed between SMN and PP results with grass most effective in reducing leached load and concentration yet cover crops most effective in reducing estimated over winter loss. However high mineralisation is likely to distort SMN balance under grass explaining this apparent discrepancy in results (Rotz et al., 2005).

Table 4-6: Summary of measurement based evaluations of mitigation in the literature.

Ref.	Details	Measurement SMN	Measurement PP	Mitigation	Main findings
Beaudoin et al., 2005	Evaluation of efficacy of Good Agricultural Practice / Agri-Environmental Practices in French catchment over 8 years	✓	✓	GAP = Cover crops, fertiliser recommendations, recycling of crop residues. AEP = GAP + Reduced fertiliser application	No significant effect mitigation level – cover crops less effective at lower level of fertilisation. Lower SMN following cover crops. Cover crops reduced mean concentration by 50% (annual basis) / 23% (rotational basis)
Beckwith et al., 1998	Field scale losses from spring barley on sandy soil in England between 1990 and 1994.	✓	✓	Manure timing Cover crop	Leached losses lower where manure applied earlier in autumn / winter. Cover crops reduced average N concentration by 74% and loads by 79% where grown on non-manured sites.
Johnson et al., 2002	Effect of 'protective' husbandry on losses in the medium term from five crop rotation on shallow limestone soil in England.		✓	'Protective husbandry' – inc. cover crops and reduced fertiliser applications	Losses and concentration standard > intermediate > protective husbandry. Impact of protective husbandry greater in second rotation than first (Johnson et al., 1997). Half rate fertiliser = 18% reduction in concentration + 30% reduction in load.
Lord et al., 1999	Field / catchment scale evaluation of Nitrate Sensitive Area measures in England between 1990 and 1996		✓	Basic = cover crops, fertiliser recommendations, manure limited to 175kg N ha-1, closed period slurry and poultry applications. Premium = Basic + arable reversion to low input grass	Cover crops = 50% reduction in nitrate leaching vs. winter crops. 40kg N ha-1 reduction in manure + 13/23% reduction in fertiliser (basic / premium area) resulted in 34% reduction leached loads + 55% reduction concentration (adjusted to local rainfall). Arable reversion = 80% reduction in loss.

Ref.	Details	Measurement SMN	PP	Mitigation	Main findings
Lord et al., 2007	Evaluation of Nitrate Vulnerable Zone Programmes of Measures in England between 2004 and 2007 at field and catchment scale	✓	✓	Current measures: Crop N requirement not exceeded	Manure N accounting = lower losses however losses greater where manure in use.
				Closed period for manure application	Elevated losses where over winter cover low or nil.
				Additional investigated measures: 10% reduction in fertiliser, removal of all manures	Modelled impact in arable catchment - reductions in leached load: Crop N requirement not exceeded = 6.5%; Closed period + crop N requirement not exceeded = 15%; 10% reduction fertiliser =7.4%; Removal all manure = 25.9%
Lord and Mitchell 1998	Effect of fertiliser inputs to cereals tested on 21 experiments on sandy soils	✓	✓	Reduced fertiliser applications	N inputs in excess of the economic optimum cause disproportionally high N loss. Reductions at supra optimal levels have greater impact on loss than reductions below the optimum.
Shepherd, 1999	Effect of cover crops on leaching from sugar beet and potatoes on sandy soils in the midlands. Conducted over 2 x four course crop rotation to investigate medium term impacts on loss.	✓	✓	Cover crops	Cover crops resulted in a 53% reduction in both leached load and concentration.
					Efficacy depended on success of previous crop (residual fertiliser), position in rotation, onset of drainage and good establishment.
Smith et al., 2002	Effect of manure application (type and timing) on N loss from grassland in England at field scale	✓	✓	Timing manure application	Significantly larger leached losses observed where slurry applied in Sept – Nov then Dec – Jun.

Analysis of farm data exposed opportunities to improve the establishment and management of cover crops to achieve larger reductions in loss. Potential cover crop uptake declines by 3.4kg N ha^{-1} per day where drilling is delayed beyond late August / early September (Vos and Van der Putten, 2004), however in some MSA / EMEL fields cover crops were not sown until mid October. The incorporation and mineralisation of cover crop residues has been reported to increase leaching above that of continuous cereals (Catt et al., 1998) highlighting the need to factor mineralisation inputs into fertiliser planning. However this was not observed in MSA / EMEL, and in some cases fertiliser inputs increased. It is also important that evaluations of cover crops / over winter state effectiveness are interpreted with respect to cropping plans. Leached loads may be lower from grass than cover crops, but its establishment represents a change in cropping rather than a modification to an existing rotation. Similarly, comparing losses from cover crops to winter crops represents a shift from winter to spring crops. Comparisons are most meaningful where made on a like for like basis i.e. spring crop vs. spring crop.

Sensitivity to rainfall highlighted the need for evaluations to be conducted over longer timescales, especially where mitigation induced change is modest. Evaluations over complete crop rotations would better account for influences of both previous and current crop on losses whilst acknowledging the prevalence of spring crops and grass within the rotation, and opportunities to plant cover crops / maintain over winter cover. The impact of mitigation is also likely to extend beyond the year of its implementation. Numerous studies investigate the impact of management and mitigation in the medium term e.g. Johnson et al., (2002). With respect to evaluations conducted in MSA / EMEL, not all mitigation methods were fully implemented before the 'mitigation winter'. Sampling ceased before fertiliser applications were made with responses to fertiliser recommendations, the most widely adopted mitigation measure, not captured in mitigation year measurements. Prior compliance with fertiliser recommendations on most MSA and EMEL farms limited scope for further reductions and highlighted the need to consider baseline conditions. More significant and consistent responses to mitigation might be expected where mitigation resulted in greater change. However the extent to which SMN and PP responses differed confirms the difficulty in addressing annual variability in short term field scale measurement.

4.4.1.2 Catchment scale measurement

Following the implementation of mitigation in 2008, N concentrations in ground / stream water decreased at 44% and 72% of sites in MSA and EMEL respectively with reductions averaging 0.9mg N l^{-1} (13%) and 1mg N l^{-1}(8%). This result suggests that high levels of leaching at the field scale, as captured by porous pots, mixed with low N flows at the catchment scale. Improvements were in keeping with those observed in Switzerland following the introduction of agri-environmental policy aimed at reducing the diffuse N loss to groundwater (Herzog et al., 2008) however improvements in Switzerland were observed at a higher proportion of sites (90%) and reflect nationwide monitoring conducted over 14 years. Given the annual variability affecting measurements, greater certainty that improvements reflect a positive response to mitigation is achieved where results are obtained over a longer period of time.

Improvements were less apparent where results were presented on a time series basis or summarised across the whole catchment. Mean concentration and crucially the number of exceedance of the drinking water standard increased post mitigation in MSA. While the opposite was true for EMEL, clear positive response to mitigation were absent in average time series of N concentration in both catchments. Coupled with a lack of consistency between the different representations of results and differences in rainfall between years, improvements observed on a site by site basis are unlikely a direct consequence of mitigation. Summer rainfall was considerably higher in 2007 than 2006 resulting in high summer leaching, as reflected in lower autumn SMN and higher N concentrations in catchment scale measurements in 2007 than 2006. Consequently N reserves were likely to have been low prior to the commencement of mitigation year sampling in autumn 2007. In addition rainfall during comparable sampling periods was 17% lower post mitigation, conducive to lower concentrations. With the reduction in rainfall exceeding improvements observed at MSA and EMEL sites, on the basis of rainfall alone, larger improvements in N concentration could have been expected. However given the complex array of processes affecting the transport and transformation of N at the catchment scale, it is not possible to confidently isolate responses to mitigation from other confounding factors not least rainfall.

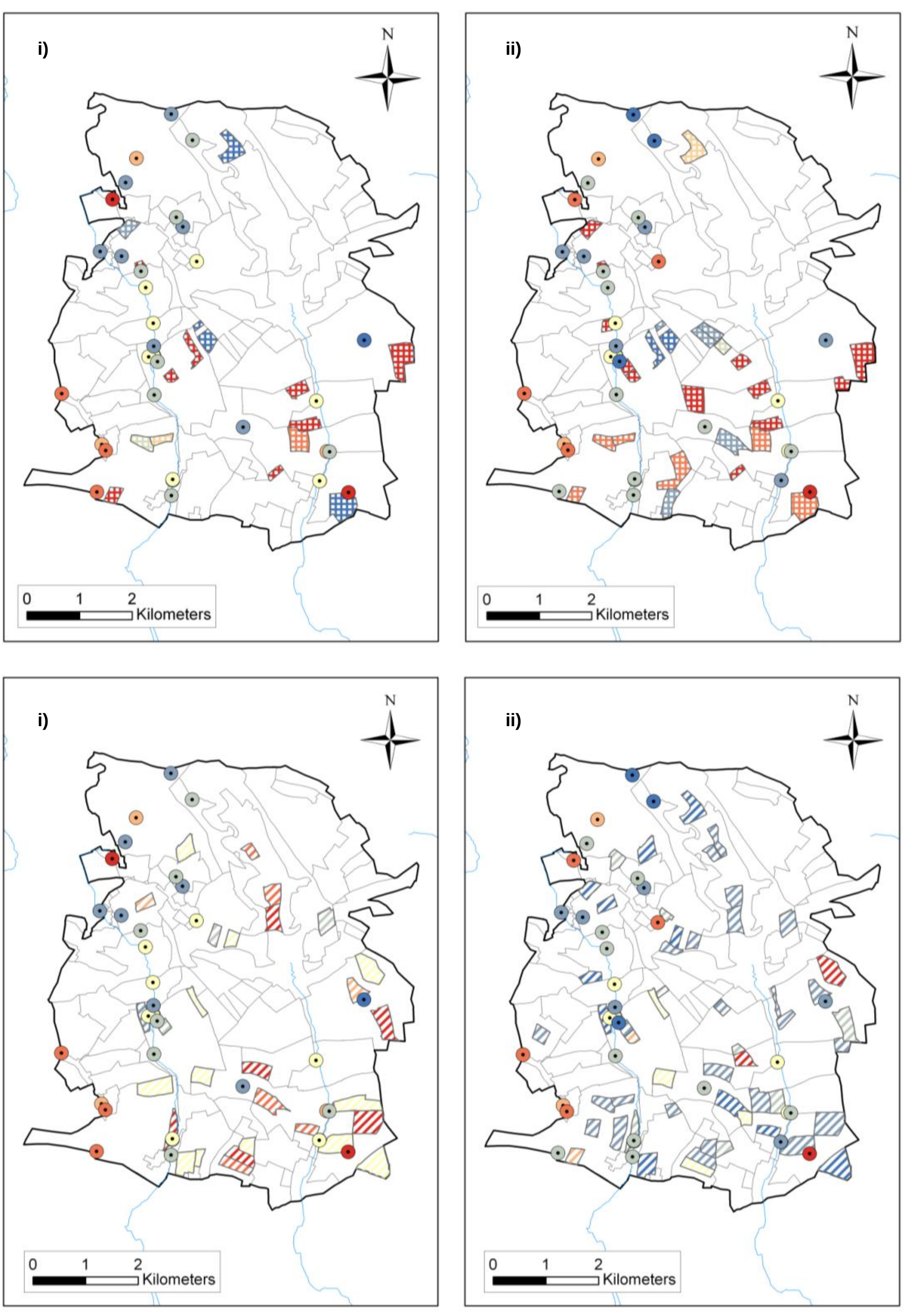
Catchment scale measurements are also affected by timelags and catchment buffering (e.g. Stalnacke et al., 2003; Granlund et al., 2005; Bechmann et al., 2008, Gutierrez and Baran 2009) which may in part explain the absence of a consistently positive response to mitigation. Reductions in N loss at the field scale take time to transpire in waterbodies especially where catchments are groundwater dominated. Silgram et al., (2005) concluded that the benefits of agricultural control schemes on groundwater quality would not be realised for several decades despite a 34% reduction in concentrations and 16% reduction in leached loads observed in short term PP measurements. The arrival of low N water is delayed by slow transit through the unsaturated zone and the need to reach abstraction depths. Although fissure flow is thought to represent the bulk of water movement in the aquifers underlying MSA and EMEL, and increased concentrations found to correspond with higher rainfall, the additional N is likely to be 'old' reflecting historic land use and long term soil surpluses (Hutchins et al., 2009). With groundwater beneath MSA and EMEL is estimated as being approximately 30 years old, and upward trends in N concentrations observed locally (Limbrick, 2003; Howden and Burt, 2008) maximum concentrations stemming from the intensification of agricultural in the 1970's may still to be reached. In addition groundwater samples tend to reflect water from a variety of depths, following different flow paths, of different age and hence different N concentration. The combination of localised flow paths with short residence times and sample sites characteristics conducive to short term responses (e.g. shallower wells / boreholes) may explain localised improvements.

4.4.2 Field vs. catchment scale measurement

Although differences in sample location prevented direct correlation of field and catchment results, N concentrations observed at the field scale were considerably larger than those at the catchment scale. PP concentrations under maize for example were c.10 times higher than the average groundwater N concentration. Differences can be attributed to the many processes and transformations affecting N beyond the soil root zone, the larger area reflected in catchment measurements and the likelihood that groundwater concentrations reflect past land use and management. Catchment measurements represent an integrated response to variability in inputs, both recent and relic, and to transformations occurring across the catchment. Spatial correspondence between PP / SMN results and catchment measurement was also low (Figure 4-18), and while this was again not surprising given the inherent differences in field and catchment scale responses, differences in sample location and low PP coverage especially in 2007 may have limited the likelihood of a more positive result. Furthermore a large proportion of catchment sampling points were located close to the catchment boundary, reflecting activities beyond the monitored area. Greater agreement was observed in terms of catchment differences with PP and groundwater concentrations both being higher in EMEL, and the degree of spatial variability greater in MSA than EMEL across all measurements approaches.

Differences in field and catchment scale responses to mitigation reflect differences in sensitivity to other factors affecting loss, namely rainfall and differences in the nature of each sampling approach (i.e. timing of measurement in relation to rainfall distribution). They also stemmed from differences in the extent of mitigation impact captured at field and catchment scale. Catchment measurements reflect the implementation of all mitigation catchment wide, and in doing so effectively account for mitigation uptake. Field scale measurements have instead demonstrated their suitability to focussed evaluations of individual field scale mitigation methods such as cover crops. However uptake must be considered in order to assess net effect on water body status. Cover crops for example were found to have a significant effect on loss on the field scale, however low uptake will have reduced impact at the catchment scale.

Sensitivity to a broader, more complex array of processes makes isolating responses to mitigation more subjective at the catchment scale and demands time series in excess of those permitted by WFD targets. However given the waterbody focus of the WFD, consideration of catchment scale processes generate more legislatively relevant evaluations and facilitates direct comparison between results and legislative standards.



PP concentration (mg N l^{-1})	0.0- 5.6 · 5.7 - 11.2 · 11.3- 19.9 · 20.0 - 29.9 · 30.0 - 39.9 · 40.0 - 60.0
Autumn SMN (kg N ha^{-1})	<50 · 50 - 99 · 100 - 149 · 150 - 199 · 200 - 249 · 250 - 299 · >300
Water quality (mg N l^{-1})	0.0 - 1.9 · 2.0 - 3.9 · 4.0 - 5.9 · 6.0 - 7.9 · 8.0 - 9.9 · 10.0 - 11.29 · >11.3

Figure 4-18: Comparison of a) PP (mg N l^{-1}) and b) SMN (kg N ha^{-1}) results with catchment scale measurement (mg N l^{-1}) in i) 2007 and ii) 2008.

4.5 Conclusions

Following the implementation of mitigation on MSA / EMEL farms, field and catchment scale measurement displayed inconsistent responses highlighting an overriding sensitivity to environmental variability when used to evaluate mitigation in the short term. Difficulty in attributing changes to mitigation was further complicated by the voluntary nature of the WAgriCo project and hence modest degree of mitigation induced change. Field and catchment scale measurements were unsuccessful in providing assessments of mitigation effectiveness where short term results were interpreted on a before vs. after basis.

Field scale measurement did however confirm the efficacy of individual mitigation methods where evaluations were performed on a 'with vs. without' basis in which assessments were made using 1 years data, thereby avoiding annual variability. Results highlighted the effectiveness of cover crops in reducing N loss with significantly lower over winter loss (estimated from SMN balances) and a tendency towards lower PP loads and concentration where spring crops were preceded by a cover crop. Field scale results also highlighted the effectiveness of grass cover in reducing losses. But despite these positive results, sensitivity to previous and current cropping, limited opportunities for mitigation within crop rotations and longer term impacts of mitigation means field scale measurement evaluations would benefit from longer term sampling irrespective of the way results are analysed.

Investigations demonstrated that catchment scale measurements are not well suited to short term evaluations of mitigation effectiveness even in catchments characterised by relatively short timelags. Results were sensitive to rainfall and their interpretation confounded by long timelags and catchment buffering. However catchment measurement captured the uptake and applicability of mitigation and coupled with its ability to evaluate mitigation at a scale of interest under the WFD, is likely to represent a useful method in the longer term. Reflecting actual change in waterbodies and providing an integrated response means fewer samples are required and can be compared directly to legislative standards. Large differences between PP and WQ results confirmed the extent of mixing and integration which occurs at the catchment scale and thus that field scale results provide only an indication of the relative magnitude of change to be expected in water body status.

5 Using field scale nutrient budgets to evaluate mitigation effectiveness

5.1 Introduction

5.1.1 Introduction to field scale nutrient budgets and their use as evaluator of mitigation effectiveness

Nutrient budgets are commonly used in agriculture to characterise nutrient management and quantify the magnitude of nutrient fluxes. Inputs and outputs to a specified system are evaluated over a defined period of time to determine whether a nutrient surplus (inputs exceed outputs) or deficit (outputs exceed inputs) exists. Where nitrogen (N) budgets are calculated, a connection is often made between nutrient surpluses and potential nutrient loss (Van Beek et al., 2003; Kyllingsbaek and Hansen, 2007; Bechmann et al., 2008); nitrogen not removed by the system is at risk of being lost to the environment. Although the relationship is sensitive to environmental and agricultural factors such as climate, soil type and land use history (Lord et al., 2002; Oenema et al., 2003), a reduction in the N surplus is likely to reduce loss and yield environmental benefits (Oenema et al., 2005). Comparing surpluses before and after the implementation of mitigation provides an opportunity to assess the effectiveness of actions to mitigation N loss to water.

Nutrient budgets can be calculated at a range of scales from plot to national scale. As the smallest subdivision on commercial farms, treated uniformly and according to site specific environmental conditions and utilisation plans, fields represent a useful scale at which to evaluate nutrient use. The soil is typically (although not exclusively) treated as the system boundary with inputs (e.g. fertiliser, manure) and outputs (e.g. crop offtake and grazing) balanced to establish whether a surplus or deficit exists. Feed → livestock→ livestock weight gain/milk/egg transfers are not inextricably linked to the field and are therefore rarely considered at this scale. Field scale calculations are commonly associated with detailed studies of nitrogen fate and transformations (e.g. Liu et al., 2003) and crop nutrient use efficiency (Webb et al., 2004), for which soil level balances offer the necessary level of detail and supporting flux measurements can be obtained. Similarly where surplus and loss relationships are being investigated (e.g. Salo and Turtola, 2006), field scale studies enable the collection of leaching / run off data with relative ease, and without the complication of connectivity issues and transformations which occur at larger scales. Field calculations provide a starting point where resources are limited, whole farms not available, or where little work has previously been conducted (e.g. Alfaro et al., 2009). Where replicates are required and experimental design is important for statistical purposes, field scale investigations afford greater control over treatments and results. Where the soil to crop transfer is of interest, field scale methodologies are well suited (Hatano et al., 2002), however their use is not limited to the field scale with soil balances (and stables balances considering the feed to milk/weight gain transfer) now included in some farm scale applications (e.g. Jurgen et al., 2006). Field scale budgets enable results to be averaged over a number of years to consider the complete crop rotation surplus, and to lessen the influence of nutrient management and residues from previous and current crops (Berry et al., 2003).

For the purpose of mitigation evaluation, fields are the entity upon which changes in farm management are realised. Decisions regarding system intensity and nutrient management may be made at the farm scale, but heterogeneity between environmental conditions and previous cropping means nutrient applications and yields are field specific. Field scale results reflect inter and intra farm variability of nutrient surpluses, highlighting 'hotspots' of nutrient excess which may result in disproportionately high loss (Van Beek et al., 2003). In doing so field budgets have the potential to inform where mitigation should be targeted, as a means to maximising cost benefit. Further, investigation at the field scale respects the non-uniform implementation of mitigation. While some mitigation methods affect all fields on a farm, for example fertiliser recommendations, other approaches are restricted to specific fields. Cover crops, for example, are applicable to spring cropped fields only, meaning field scale evaluation of more appropriate for some mitigation options. Conversely, some methods such as changes in livestock feeding regimes are implemented at the stall level, and have no direct link to field management. Thus in some instances farm scale evaluations may be more appropriate. For this reason Chapter 6 investigates the usefulness of farm scale nutrient budget assessments, while chapter 8 compares the two approaches.

Given the waterbody focus of the Water Framework Directive (WFD), the applicability of assessment methods at the catchment scale must also be considered. Results obtained at the field scale can be aggregated across catchments to assess wider impact (e.g. Ruiz et al., 2002). The underlying field scale data provides detailed catchment assessments, albeit with more intensive data handling and manipulation. Where investigations are conducted at the regional scale and/or over the longer term, agricultural census data may replace field scale input data. By adopting lower resolution input data, soil level budgets can be utilised at much larger scales without excessive computation (Jansons et al., 2003; Wendlund et al., 2005).

Given the applicability and relative practical ease of obtaining field scale measurements using SMN and PP techniques, few examples of field scale budget mitigation evaluations exist. However, with nutrient management and mitigation implemented at the field scale, and budgets inherently sensitive to changes in inputs and outputs, field scale budgets may possess an as yet unrealised potential for mitigation evaluation. This chapter aims to redress the absence of field scale mitigation evaluations and explore their usefulness as a means of assessing mitigation effectiveness. Although the adoption of improved practices and mitigation is likely to affect surpluses, the extent and speed to which these changes might transpire in budgets has not been reported. The study aimed to investigate the sensitivity of field scale nutrient budgets to a range of mitigation methods, adopted both individually and in combination. With the WFD demanding improvements in water body status, exploring catchment scale impact of mitigation is an important requirement of monitoring methods. The option to upscale field scale results and provide an indication of catchment impact was therefore included as an additional investigative component. With these intentions in mind, a hypothesis, aim and associated objectives are proposed in section 5.1.2.

5.1.2 The current study

5.1.2.1 Hypothesis 1

‘Field scale nutrient budgets represent an effective method of mitigation evaluation at field and catchment scale.’

5.1.2.2 Aim

To assess the sensitivity of field scale nutrient budgets to a range of mitigation methods (observed and simulated) at field and catchment scale.

5.1.2.3 Objectives

1. To develop an appropriate field scale nutrient budget methodology for the purpose of mitigation evaluation.
2. To explore field scale nutrient management and spatial variability.
3. To investigate the impact of mitigation on nutrient surpluses by:
    a. Comparing surpluses before and after the implementation of mitigation for a representative subset of MSA and EMEL fields.
    b. Simulating mitigation scenarios to increase the range and magnitude of mitigation employed.
4. To upscale field scale results to the catchment scale.

5.2 Method

5.2.1 Field scale budget methodologies

Field scale nutrient budgets are calculated almost exclusively using a soil→crop transfer budget which treat the soil as the system boundary (Schorder et al., 2003). Varying degrees of complexity can be adopted depending on the extent and purpose of the study, and the availability of input data (Oborn et al., 2003; Oenema et al., 2005). The simplest approach is that of a soil surface balance which considers nutrients entering and leaving a field via the soil surface, but does not consider any internal cycling of N (Figure 5-1a); the balance between mineralisation and immobilisation is therefore not considered. Where a more detailed understanding of nutrient fate is required, soil system methodologies are used which include mineralisation inputs, and apportion N losses to leaching, run off, denitrification and ammonia volatilisation (Figure 5-1b). While soil system budgets provide a more comprehensive representation of the soil system, there is increased dependence on site specific measurements (Oborn et al., 2003). Soil system balances are typically used for research and development rather than management and policy, and are unsuitable for extensive projects (Oenema et al., 2003). Soil surface budgets are more common in the literature, however variation in the inputs / outputs considered (Table 5-1) means surpluses must be interpreted with respect to the methodology in use, respecting loss processes already accounted for (Oenema et al., 2005).

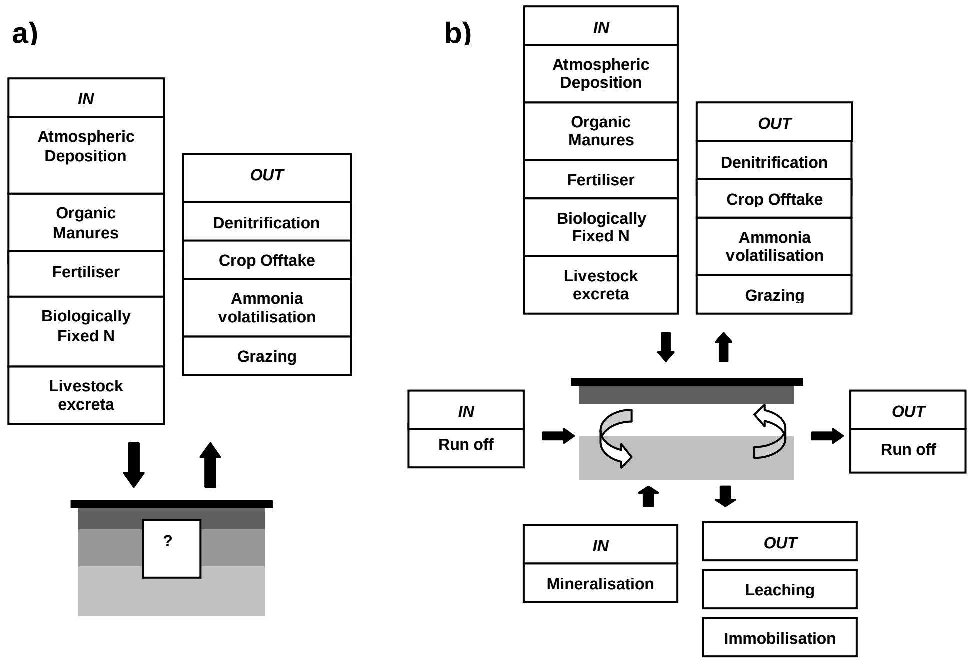


Figure 5-1: Nutrient fluxes accounted for by a) soil surface and b) soil system budgets (adapted from Oenema et al., 2003)

5.2.2 Developing a field scale methodology

To investigate the sensitivity of field scale nutrient budgets to mitigation methods, a suitable nutrient budget methodology was required. In agreement with Oenema et al., (2005), the purpose of the study determined the type of budget in use; however decisions were constrained by data availability. The aim was to develop a comprehensive methodology to assess mitigation effectiveness that matched input data availability. Utilisation of balance results for the investigation of surplus – loss relationships (see chapter 8) placed additional restrictions on the evolving methodology. Deviation from existing methodologies was accepted given the unique functionality of the budget produced.

Available field input data was identified (Table 5-1) and compared to inputs / outputs included in existing methodologies (Table 5-2). The magnitude of fluxes was quantified and their relative contributions assessed (Table 5-3). Table 5-1, Table 5-2 and Table 5-3 were compared to facilitate selection and justification of balance components. Initially inputs / outputs contributing most to the balance were selected (with particular emphasis on those obtained from field data). Where justifications and the availability of input data did not correspond, the option of using standard values from the literature was also considered. To maximise sensitivity to mitigation, emphasis was placed on inputs affected by nutrient management, and loss processes represented by standard, fixed values avoided.

5.2.2.1 Availability of input data

Field scale farm data was obtained from the WAgriCo project. Farmers supplied data from 2005 – 2008 during one to one meetings with an assigned catchment advisor. Whole farm data regarding livestock numbers, livestock movement and the import / export of feed, bedding, manures, fertiliser, animal products and crops were obtained in addition to field specific cropping and management information. Table 5-1 details data collected at the field scale and comments on its availability / quality.

5.2.2.2 Budget components

The following inputs / outputs have been identified in existing methodologies. Their inclusion / exclusion in a mitigation assessment balance is now discussed with reference to Table 5-1, Table 5-2, and Table 5-3.

INPUTS

Fertiliser: Fertiliser represents the largest nutrient input on most conventional farms (especially on arable farms), and is therefore included in almost all balances. Information regarding field fertiliser applications is available for the majority of MSA/EMEL fields, and the quality of this data is high due to NVZ record keeping requirements and WAgriCo data exchange agreements. Fertiliser applications are commonly targeted by mitigation methods further justifying their inclusion in the balance.

Manure / excreta: On dairy and intensive livestock farms, on-farm excreta production is high, yielding large quantities of organic manure deposited directly to grass during grazing or collected in housing and subsequently spread onto fields. Due to high fertiliser prices and limits on maximum manure application rates, manures are also exported from dairy / livestock farms and imported onto arable farms. On organic farms, manures are relied upon to maintain soil fertility. Although the value of manure is increasingly appreciated, excessive and untimely manure applications still occur (Chambers et al., 2000). The potential to improve manure management is considerable and the target of many mitigation methods (Cuttle et al., 2006). Although manure related farm data was in places unreliable, the large quantities applied and relevance for mitigation necessitates its inclusion.

Table 5-1: Field scale farm data collected during the WAgriCo Project (2005 – 2008)

Frequency	Data field	Availability	N input / output
Once	Field size	High	
	Field slope	Low[1]	
Annually	Harvest year	n/a	
	Crop	High	
	Cropped area	Medium[2]	
	Drill date	Medium	
	Harvest date	Medium	
	Yield or number cuts	Medium[3]	✓
	Residue fate	Medium	✓
	Sub-soiled date	n/a[4]	
	Cultivation type	Low	
	Cultivation date	Low	
	Manure type	Medium	
	Manure application rate	Medium	✓
	Manure application date	Medium	
	Manure incorporation	Medium	
	Fertiliser application rate	High	✓
	Fertiliser application date	High	
	Grazing details	Medium[5]	✓

[1]Supplemented by additional slope measurements taken by WAgriCo staff

[2]Where no detail, assumed cropped area = field size

[3]Details of grass cutting less complete than yield information

[4]Only 8 fields sub-soiled during the project

[5]Details of grazing limited

Table 5-2: Existing field scale nutrient budget methodologies

Reference	Country	Inputs							Outputs					
		Mineral fertiliser	**Organic manures / grazing excreta**	**N fixation**	**Atmos. Depos.**	**Min.**	**Run off**	**Seed**	**Crop offtake inc. grazing**	**Ammonia volat.**	**Denitrif.**	**Leaching**	**Run off**	**Comments**
Alfaro et al., 2009	Chile	x	x	x	x	x	X (rainfall)		x	x		x		Also calculated a field gate balance
Liu et al., 2003	China	x				x			x	x	x	x		Also included initial soil nitrate-N before planting (input) and residual soil profile nitrate-N (output)
Berry et al., 2003	UK		x	x	x			x	x	x		x		Organic farms therefore no mineral fertiliser input. Also considered animal products due to grazing (as an additional output not inc in crop offtake)
Hatano et al., 2002	China	x	x	x	x				x	x	x			Ammonia volatilisation losses from manure application
MIDaS project (ADAS, 1999)	UK	x	x		x				x					
Salo and Turtola, 2006	Finland	x	x	x					x	x				

Table 5-3: Comparison of flux magnitudes (kg N ha^{-1}). Unless stated otherwise, table show mean (and range) of values in literature. For inputs: manure (max figure) > N fixation > fertiliser (average of arable and grassland) > mineralisation > atmospheric deposition. For outputs: leaching > ammonia volatilisation > denitrification. See table A-1 (in the appendix) for full details.

Input / output	Flux	Magnitude
Inputs	Atmospheric deposition	35 (30-40) kg N ha^{-1}
	Fertiliser [a]	Arable = 147, Grassland = 69 kg N ha^{-1}
	Excreta [b]	170 kg N ha^{-1}
	Manure [b]	
	Mineralisation	66 (26-115) kg N ha^{-1}
	N fixation	139 (5-300) kg N ha^{-1}
Outputs	Crop offtake / grazing	Arable = 144, Grassland =233 kg N ha^{-1} [c].
	Ammonia volatilisation	8% (1.1-12.6%) excreted N, 24% (22.0-26.0) of manure N in housing, 15% (6.1-20.9%) stored manure N, 36% (9.0-48.6%) spread manure N [d].
	Denitrification	3% (2-4%) excreta N, 6% (2-16.2%) fertiliser N, 4% (1.25-12.6%) manure N [d]
	Leaching	Arable = 96 (10.3-306.4) v, Grass= 85 (30- >200) kg N ha^{-1}

[a] Average England and Wales fertiliser application to arable / grassland between 2005-2008 (BSFP, 2009)

[b] NVZ maximum total N application of manure inclusive of excreta

[c] Average of WAgriCo crop offtake

[d] % TN (total N)

Although manure application data was available for each field, information regarding grazing practices and subsequent field receipts of excreta were provided on a generalised farm wide basis. Although details of livestock excreta production and N content are well documented, information regarding stocking density and grazing period for individual fields was not obtained. A method of distributing excreta between fields was therefore developed (see section 5.2.2.3)

Atmospheric deposition: Atmospheric deposition describes the transfer of atmospheric nitrogen to land. It is commonly included in existing methodologies, and represents a significant contribution to agricultural systems. Although atmospheric inputs have not been measured locally, the use of a fixed input is common (e.g. Berry et al., 2003). Owing to the acceptance of this approach, the availability of input values and the consistent contribution to all fields, inclusion was justified. A fixed value of 30kg N ha^{-1} (Goulding et al., 1998) was adopted.

N fixation: Nitrogen fixation refers to the direct attainment of nitrogen from the atmosphere by organisms. The process is facilitated by symbiotic micro-organisms (e.g. rhizobia associated with the roots of leguminous crops). Despite inclusion in a number of balances, it was decided that N fixation would not be included on this occasion. Inconsistent assessments of clover coverage and low catchment coverage of leguminous crops meant inclusion would be neither significant nor reliable.

Mineralisation: Mineralisation refers to the decomposition and associated release of nitrogen from organic matter (including manure and residues). Nitrogen is released as ammonium-N which is subsequently nitrified to nitrate-N. As a soil system balance rather than a soil surface component, mineralisation is less common as a budget input. However soil organic matter is the largest pool of nitrogen in the soil, and where conditions are appropriate (warm, wet and low C:N ratio), mineralisation inputs can be large (Shepherd et al., 1996). Mineralisation inputs tend to be larger where regular manure applications have been made (Shepherd, 1993). Manure related mitigation is likely to affect mineralisation inputs.

The release of N via mineralisation is a continual and dynamic process, with different fractions decomposing at different rates (Chambers et al., 1999; Ledgard et al., 1999; Lord et al., 2007). With balances typically calculated on an annual basis, multiple measurements are required to capture the complex and transient balance between mineralisation, immobilisation and crop demand (Withers and Lord, 2002). Limited resources meant mineralisation inputs could not be measured directly. Autumn soil mineral nitrogen (SMN) measurements were however available for a subset of fields, quantifying total ammonium and nitrate N, the end products of mineralisation; these results acted as a proxy for mineralisation inputs. However they represent only as a snapshot, and when measured early in the new harvest year (budgets are in calculated across a harvest year), they provide no insight into mineralisation inputs during the rest of the year.

With SMN results available for some fields it was decided that two budgets would be calculated; a soil surface balance, and a soil system balance. In doing so any uncertainties introduced by the use of an over simplified mineralisation input would not be extended to all results, and there would be an opportunity to compare the merits of the two approaches.

Run off: Runoff is typically a feature of soil system not soil surface budgets. It is infrequently included and there is a lack of data quantifying likely contributions. With no local measurements obtained, it was excluded from both budgets.

Seed: Although a relatively straightforward input to model, seeds represent a small nutrient input, and are unlikely to be the target of mitigation methods. A seed component was therefore excluded from the balance to reduce computation.

OUTPUTS

Crop offtake / grazing: In productive agricultural systems, crop offtake and grazing represent the largest nutrient output and are therefore included in the majority of budget methodologies. Although mitigation methods do target offtake, inclusion provided an opportunity to explore sensitivity to environmental conditions and to test the robustness of a field budget as an assessment method. While yield data was available for most fields between 2005 and 2007, a late harvest and completion of the WAgriCo project meant some 2008 values were estimated by catchment advisors. In spite of this, the relative magnitude of offtake justifies its inclusion.

While calculating arable offtake is straightforward, distributing livestock between fields given the limited grazing detail is more complex. A means of dividing Livestock Units (LU) between fields based on the number of grazing defoliations was subsequently devised (see section 5.2.2.3).

Ammonia volatilisation: Ammonia volatilisation describes the direct loss of ammonia gas from manures and livestock excreta. Losses are dependent on the total ammoniacal N content and manure management practices. Agriculture is responsible for 90% of the UK ammonia emissions (Misselbrook et al., 2000); the significance of losses is reflected in the common inclusion of ammonia volatilisation in budget calculations. Although ammonia fluxes were not measured locally, losses were estimated using an 'accountability factor' which determines the proportion of total manure N lost during housing, storage and spreading based on manure type, environmental conditions and incorporation delay. Accountability factors were produced using the Manure Nitrogen Evaluation Routine (MANNER) (Chambers et al., 1999) software and data from Defra project WT0715NVZ (Smith and Cottrill, 2007). Further details regarding their development can be found in chapter 7.

Table 5-4: The complete field scale budget – Inputs

INPUT (kg N ha^{-1})	**Calculation details**	**References**
Total Fertiliser	Σ [Fertiliser application rate (kg N ha^{-1}) x Fertiliser N content (% N) / 100]	
Total net manure N	Total gross manure N (kg N ha^{-1}) x Accountability factor [a]	See section for details of the development of accountability factors
	Total gross manure = Σ [Manure application rate (t ha^{-1}) x Manure N content (kg N t^{-1}]	Manure N contents from RB209 (Defra, 2000) and PLANET V2 (ADAS, 2008)
Total net field excreta input	Total gross field excreta input (kg N ha-1)[b] x grazing accountability factor (0.95)	Grazing accountability factor
	Total gross field excreta input = Total grazing excreta (kg N) / total grazing defoliations (defoliation ha's) x number field defoliations	
	Total grazing excreta (kg N) = Total livestock excreta (kg N) x Proportion of year grazed.	
	Total farm livestock excreta (kg N) = Σ [Livestock annual excreta rate (kg N animal^{-1}) x No. animals]	Livestock excreta N values from Defra project WT0715NVZ (Smith and Cottrill, 2007)
	Total grazing defoliations (defoliation ha's) = Σ [Field area (ha) x No. grazing defoliations]	
Atmospheric deposition	30 kg N ha^{-1}	Goulding et al., 1998
Mineralisation	Total SMN (kg N ha^{-1})	Local soil mineral nitrogen sampling

[a] Ammonia losses were considered an input reducer not a separate loss pathway because they are a function of the total manure input.

[b] In the absence of detailed grazing information, excreta was estimated on a 'per defoliation' basis; the more grazing occassions each field is subject to, the more excreta deposited. A number of assumptions were made:

- Grazing defoliations are of equal length.
- Excreta inputs were distributed evenly across each field
- Stocking densities are consistent across the farm and year – full account is therefore taken of differences in field size.

Stocking densities may differ between fields (higher in smaller fields where the herd is kept together) and be compensated by shorted grazing periods. Total excreta N was then apportioned in accordance with the number grazing defoliations and the length of the grazing season.

Denitrification: Denitrification describes the chemical reduction of nitrate and nitrite to nitrous oxide gas under anaerobic conditions. It is less commonly included in nutrient budgets although under certain conditions losses can be quite large (Conon et al., 2000). Although not measured locally, values from the literature are available. However with mitigation likely to have an indirect impact on denitrification, the use of fixed values would be inappropriate and reduce the sensitivity of surpluses to mitigation. Further, the adoption of standard losses would ignore possible pollution swopping whereby leaching losses are reduced at the expense of nitrous oxide emissions. Given the clay loam, moderate organic content and low water content soil conditions observed within the study catchments, denitrification losses were not expected to be high.

Table 5-5: The complete field scale budget – Outputs

OUTPUT (kg N ha^{-1})	Calculation details	References
Total arable offtake	Total grain N (kg N ha^{-1}) + Total removed straw N (kg N ha^{-1})	
	Total grain N (kg N ha^{-1}) = yield (t ha^{-1}) x Grain N content (kg N t fw^{-1})	Grain and straw N contents (Sylvester-Bradley, 1993) / PLANET v2 (ADAS, 2008). Where no yield information was available, values were estimated by catchment advisors / based on past farm yields and inter-annual trends.
	Total straw N (kg N ha^{-1}) = yield (t ha^{-1}) x 0.65 x straw N content (kg N t fw^{-1})	Grain: straw ratio 1:0.65 (Lloyd, 2008 pers. comm. - WAgriCo catchment advisor)
Total grass offtake	Total cut grass (kg N ha^{-1}) + Total grazed grass [a] (kg N ha^{-1})	Grass N content from PLANET v2 (ADAS, 2008)
	Total cut grass = Grass N content (kg N t fw^{-1}) x silage yield (t ha^{-1})	Where no silage yield information was available, standard values from RB209 (Defra, 2000) were used.
	Field grazed offtake = Total grazed offtake (kg N ha^{-1}) / total farm defoliation areas x number field defoliations	Grazing rate figures from Hopkins (2000)
	Total grazed offtake (kg N ha^{-1}) = Σ [Livestock units (dairy / beef / sheep) (LU's) x grazing rate (kg DM LU^{-1} day^{-1}) x grazing days (days) x grass N content (kg N t^{-1} @ 100% DM)]	
	Total grazing defoliations (defoliation ha's) = Σ [Field area (ha) x No. grazing defoliations]	

[a] For consistency with excreta inputs, grazing outputs were also estimated using 'defoliation areas' i.e. the number times grass was grazed. Total annual grazing grass N was again apportioned according to the field specific number of defoliations.

Leaching: Leaching describes the downward movement of nitrate-N through the soil profile. Although leaching losses are smaller than those associated with crop offtake, the environmental and economic consequences are far more significant. Leaching losses were measured in a subset of representative fields, and estimated values available from models such as IRRIGUIDE (Bailey and Spackman, 1996) and NITCAT (Lord, 1992). However, as the target of mitigation, it was important leaching remained part of the surplus. Adopting fixed/modelled values would significantly reduce sensitivity to mitigation.

5.2.2.3 Calculating the budget

Having considered the benefit of including each input / output, two possible methodologies were constructed, a soil surface balance (excluding mineralisation) and a soil system balance (including mineralisation). The final set of inputs and outputs, associated calculations and the sources of input data are summarised in Tables 5-4 and 5-5. The budget was calculated annually on a 'per ha' basis by subtracting total output from total input. Worked examples for arable and grassland fields are provided in Table 5-6 and Table 5-7 respectively.

Table 5-6: Worked example of field scale budget calculation for an arable field

Budget component	**Farm data**	**Calculation**		**kg N ha^{-1}**
INPUTS				
Fertiliser	120 kg N ha^{-1} fertiliser	Σ [Fertiliser application rate (kg N ha-^{1}) x Fertiliser N content (% N) / 100]	120 x 100[a] / 100	120.00
Net manure	10t ha^{-1} layer manure incorporated within 12-24hours	[Manure application rate (t ha^{-1}) x Manure N content (kg N t^{-1}] x Accountability factor	10 x 16[b] x 0.51[c]	81.60
Atmospheric deposition	n/a – Std. value	n/a		30.00
Mineralisation	SMN measurement	n/a		293.20
TOTAL	Soil surface balance			231.60
	Soil system balance			524.80
OUPUTS				
Grain	8.6t ha^{-1} WWF, straw baled and removed	Yield (t ha^{-1}) x Grain N content (kg N t fw^{-1})	8.6 x 17[d]	146.20
Straw	Baled and removed	Yield (t ha^{-1}) x 0.65 x straw N content (kg N t fw^{-1})	8.6 x 0.65 x 5[e]	27.95
TOTAL				174.15
BALANCE	Soil surface balance			57.45
	Soil system balance			350.65

[a] Fertiliser application given as total N therefore N content is 100%

[b] Layer manure N content = 16 kg N t^{-1}

[c] Accountability factor for layer manure incorporated within 24hours = 0.51

[d] WWF grain N content = 17 kg N t fw^{-1}

[e] WWF straw N content = 5 kg N t fw^{1}

Table 5-7: Worked example of field scale budget calculation for a grass field

Budget Component	Farm data	Calculation		kg N ha^{1}
INPUTS				
Fertiliser	170 kg N ha^{-1} fertiliser	Σ [Fertiliser application rate (kg N ha-^{1}) x Fertiliser N content (% N) / 100]	170 x 100[a] / 100	170.00
Net manure	42t ha^{-1} slurry not incorporated	[Manure application rate (t ha^{-1}) x Manure N content (kg N t^{-1}] x Accountability factor	42 x 3[b] x 0.73[c]	91.98
Net excreta	300 dairy cows, 70 growers and 200 calves on farm. 249.52[d] defoliation ha's on farm and 2 grazing defoliations on field. 6 months grazing per year	Total farm livestock excreta (kg N) x proportion of year grazed / total grazing defoliations (defoliation ha's) x number field defoliations x grazing accountability factor	38294.34[e] x 0.5 / 249.52[d] x 2 x 0.95	145.80
Atmospheric deposition	n/a – Std. Value	n/a		30.00
Mineralisation	SMN measurement	n/a		171.20
TOTAL	Soil surface balance			437.78
	Soil system balance			608.98
OUPUTS				
Grazing	300 dairy cows, 70 growers and 200 calves on farm. 249.52[d] defoliation ha's areas on farm and 2 grazing defoliations on field. 6 months grazing per year	Total farm grazed offtake (kg N ha^{-1}) / total farm defoliation areas x number field defoliations	29362.42[f] / 249.52 x 2	235.35
Silage	10.4 t fw ha^{-1} silage	Grass N content (kg N t fw^{-1}) x silage yield (t ha^{-1})	6.8[g] x 10.4	70.72
TOTAL				306.07
BALANCE	Soil surface balance			131.71
	Soil system balance			302.91

[a] Fertiliser application given as total N therefore N content is 100%

[b] Dairy slurry N content = 3 kg N t^{-1}

[c] Accountability factor for cattle manure not inc = 0.51

[d] Total grazing defoliations (defoliation ha's) = Σ [Field area (ha) x No. grazing defoliations] = (83.32 x 1) + (83.1 x 2) = 249.52 defoliation ha's

[e] Total farm excreta = Σ [Livestock annual excreta rate (kg N $animal^{-1}$) x No. animals] = (300 dairy cows x 117) + (10 beef sucklers x 68) + (63 growers x 38) + (200 calves x 9.3 x 0.058 (on farm for 3 weeks only) = 38294.34 kg N

[f] Total farm grazed offtake = Σ [Livestock units (dairy / beef / sheep) (LU's) x grazing rate (kg DM LU^{-1} day^{-1}) x grazing days (days) x grass N content (kg N t^{-1} @ 100% DM)] = (360 dairy LU x 15 kg DM LU^{-1} day^{-1} x 180 x 27.2 / 1000)+(49.77 beef LU x 12 kg DM LU^{-1} day^{-1} x 180 x 27.2 / 1000) = 29362.42 kg N

[g] Grass N content = 6.8 kg N t fw^{-1}

5.2.3 Calculating field scale budgets

5.2.3.1 Field selection

With over 500 fields in the MSA and EMEL catchments, field budgets were calculated for a subset of fields. Field selection focussed on three criteria:

1. Representative cropping
2. SMN / PP monitoring (for further investigation regarding relationships between surpluses and loss – see chapter 8).
3. Fields subject to field level mitigation, namely cover crops.

For MSA all 'original' SMN and PP fields were selected (those selected in 2007 and not including those added in 2008). Additional mitigation fields were also added. Careful selection of fields for monitoring meant the subset was approximately representative of catchment cropping. The availability of field manure data affected EMEL field selection and only those fields with sufficient data were considered. As a result the EMEL subset contains fewer fields than MSA. Selection was initially based on fulfilment of criteria 2 and 3. Additional non mitigation fields were added to ensure the subset was representative of cropping and farms.

A total of 84 fields were selected (60 in MSA and 24 in EMEL) covering 25% and 5% of MSA and EMEL by area respectively. In both catchments there was some bias towards spring cropped fields, and against grass fields (Table 5-8) relative to cropping across each catchment (see Figure 3-9). Spring barley and maize fields were over-represented to maximise the number of cover cropped fields and more fully investigate the impact of mitigation. Conversely mitigation was expected to have less impact on grass fields where fertiliser applications were known to be lower than recommended values, and with a limit to the size of the subset, there was a need to ensure other crop types were sufficiently represented to ensure reliable results across all crop types.

5.2.3.2 Data Interpretation

Soil surface budgets were calculated on an annual basis between 2005 and 2008. Where SMN data was available, soil system balances were also calculated. Investigations focused on eight dominant crops (grass, maize, SBM, WBF, WO, WOSR, WWF and WWM) which occupied c.90% of the catchment in each year (see Figure 3-9).

To investigate sensitivity to mitigation, pre- (2005 – 2007) and post- mitigation (2008) results were compared. With crop rotations exceeding the 4 year study period, and crops typically changing annually, it was not possible to calculate budgets across a complete crop rotation on individual fields. Instead results from 2005 – 2007 from all fields were averaged to produce crop specific 'before' results. All 2008 results were classified as 'after' results on the grounds that fertiliser recommendations were, as a minimum, adhered to on all fields. Results were excluded from analysis where any input / output data was missing.

Comparisons of before and after results assessed the cumulative response to all mitigation. To investigate the impact of individual mitigation methods, results from fields with / without specific methods were compared, and changes in nutrient use on individual farms explored. Farm specific investigations focused on WWF, SBM and maize fields due to high coverage / suitability for mitigation, and were restricted to farms supplying results across all four years. For each crop changes in nutrient use were identified and attributed to individual mitigation methods.

5.2.3.3 Data analysis

Results were analysed using GENSTAT v12. General ANOVAs were performed to test for significant differences in inputs, outputs and surpluses before vs. after the implementation of mitigation and with vs. without specific mitigation methods. Due to similarities in the factors likely to explain variation in results (e.g. crop type), analysis followed a very similar format to that adopted for field scale measurement results. The reader is therefore directed to section 4.2.3.3 for more details.

5.3 Results

5.3.1 Effect of mitigation on field scale budgets

Soil surface surpluses tended to decrease following the implementation of mitigation, however only in EMEL were differences between before and after results significant (Figure 5-2 and Figure 5-3). Maximum reductions were associated with maize and WWF fields in both catchments whilst maximum increases were observed for WOSR. Surplus reductions in EMEL were driven by a significant reduction in total inputs stemming from significantly lower manure to all crops except WOSR. For grass, maize and WWF fertiliser inputs were also lower after mitigation but differences before and after mitigation were not significant (Table 5-8). In contrast the initial downward trends observed in MSA resulted from significantly higher total output, driven by significantly higher yield post mitigation. A significant reduction in fertiliser inputs to all MSA crops except WO and WOSR was counteracted by increased manure inputs.

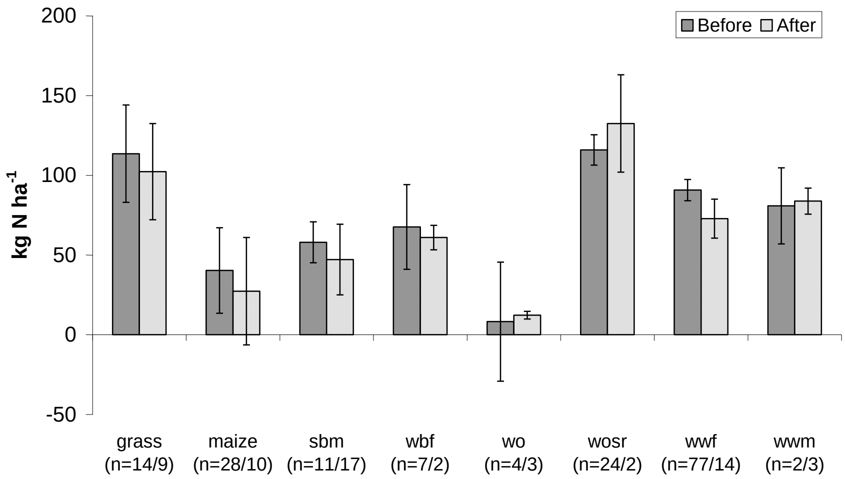


Figure 5-2: Soil surface surpluses in MSA before and after the implementation of mitigation. Mean and standard error shown with sample size before and after mitigation in parentheses. Differences between before and after surpluses not significant to p<0.05. See Table 5-8 for full details of statistical analysis.

Responses to mitigation were not significantly different between MSA and EMEL surpluses with no significant interaction between catchment and year (Table 5-8), however fertiliser and manure inputs and yield were significantly different between catchments; prior to mitigation fertiliser and manure use was consistently higher in EMEL. Yield was also generally higher in EMEL however differences in residue management between catchments (i.e. the fate of straw) meant differences did not transpire in total outputs which were similar between catchment. In terms of surpluses, total inputs and totals outputs where not sufficiently different to yield significant differences in surpluses between catchments.

Fertiliser, manure, total inputs, yield and total output were significantly different between crops in both catchments confirming the crop specificity of nutrient inputs and offtake. Fertiliser inputs were largest to grass and winter wheat whilst manure inputs were largest to grass and maize. Total inputs were largest on grass fields which also received livestock excreta during grazing. In terms of output, highest offtake was observed on grass, maize and winter wheat fields. High offtake relative to inputs meant surpluses were smallest for cereal crops. High inputs / low offtake meant surpluses were largest for grass / WOSR.

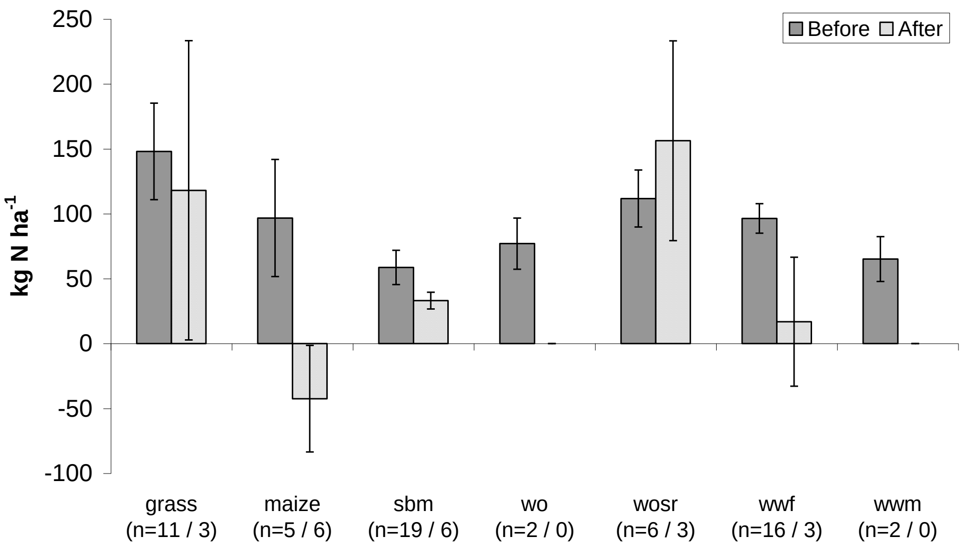


Figure 5-3: Soil surface surpluses in EMEL before and after the implementation of mitigation. Mean and standard error shown with sample size before and after mitigation in parentheses. Differences between before and after surpluses significant to p<0.05. See Table 5-8 for full details of statistical analysis.

To account for annual variability in nutrient use driven by the weather and changes in input price, results were also analysed on an annual basis and compared to national trends. In doing so differences before and after mitigation could be interpreted more objectively and change in nutrient use more accurately attributed to mitigation. Neither fertiliser nor yield were significantly different between individual years (Figure 5-4 and Figure 5-5), resulting in no significant differences between annual surpluses. However over the four year study period, national fertiliser use decreased (Figure 5-4), in part due to increasing price. A similar downward trend across all years was less evident in MSA and EMEL however largest reductions in fertiliser use were not always observed between 2007 and 2008 coinciding with the implementation of mitigation. It is also worth noting that fertiliser use was generally higher in MSA and EMEL than observed nationally. This was despite a larger proportion of farms in MSA and EMEL using manure than observed nationwide (fertiliser use would be expected to be lower where manure is used) – see note in Figure 5-4. Trends in national average yield between years were less apparent although in keeping with higher fertiliser use in MSA and EMEL yields were higher in MSA and EMEL than across the UK. However as noted above, MSA and EMEL yields tended to be higher after mitigation, and in MSA, improvements in surpluses were predominately driven by higher output not lower inputs.

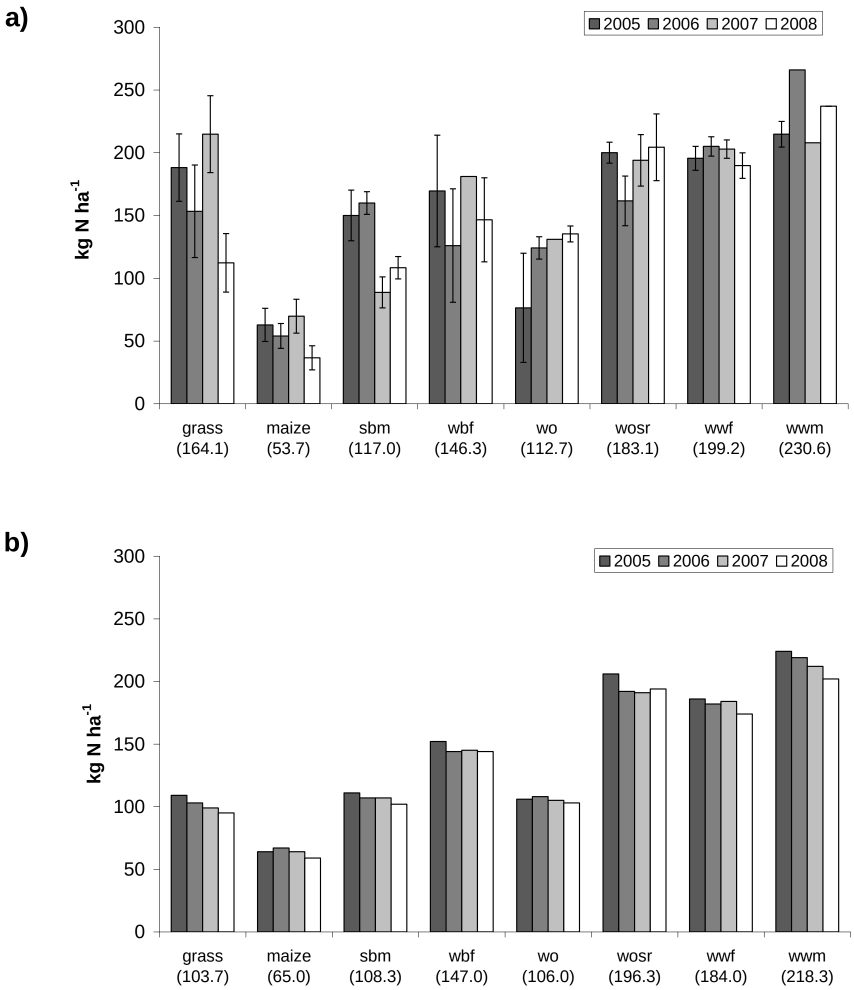


Figure 5-4: Average fertiliser use (kg N ha^{-1}) in (a) MSA/EMEL and (b) UK (BSFP, 2009) between 2005 and 2008. Differences between years in MSA/EMEL not significant to p<0.05. Application rates averaged across all fields including those receiving manure. The proportion of fields receiving manure in MSA/EMEL is higher than that nationally; in 2008 the proportion of farms with at least one field receiving manure was 69 and 88% for the UK and MSA/EMEL respectively. Application rates would be expected to be lower where manure is also applied. 2005-2008 average shown in parentheses for comparison between UK and MSA / EMEL.

Table 5-8: Selected components of the soil surface balance in MSA and EMEL before and after mitigation with results of ANOVA analysis – p values denote the significance of differences and interactions between crop type / year /catchment. Manure presented on a gross (before ammonia loss) basis. Yield is not a direct balance output but included to explain changes in total output. Recommended fertiliser application rates also shown to highlight scope for reduction in inputs. Comparison should be made against fertiliser and fertiliser+manure because total input = fertiliser + manure (net of ammonia loss) + excreta + atmospheric deposition. Total output = grain + straw offtake.

Catchment	Crop	n		Fertiliser (kg N ha^{-1})		Manure (kg N ha^{-1})		(RB209 Fert. Rec). (kg N ha^{-1})	Total inputs (kg N ha^{-1})		(Yield) (T ha^{-1})		Total output (kg N ha^{-1})		Surplus (kg N ha^{-1} / %)		
		Before	*After*	*Before*	*After*	*Before*	*After*	-	*Before*	*After*	*Before*	*After*	*Before*	*After*	*Before*	*After*	%
MSA	Grass	14	9	173	84	93	98		375	281			262	179	114	102	9.9
	Maize	28	10	61	33	197	225	80	235	208	40.5	37.7	195	181	40	27	32.3
	SBM	11	17	108	99	39	70	120	159	171	5.8	7.2	101	123	58	47	18.7
	WBF	7	2	146	147	50	99	200	207	227	6.9	8.0	139	166	68	61	9.8
	WO	4	3	83	135	0	0	130	113	165	5.9	7.9	104	153	8	12	-49.0
	WOSR	24	2	176	195	40	68	220	225	257	3.8	4.2	109	125	116	132	-14.3
	WWF	77	14	197	194	47	50	240	252	253	8.3	9.0	161	180	91	73	19.7
	WWM	2	3	246	237	0	11	240	276	275	8.8	8.6	195	191	81	84	-3.7
	Crop	*n/a*		*P<0.001*		*P<0.001*		*n/a*	*P<0.001*		*P<0.001*		*P<0.001*		*P<0.001*		
	Year	*n/a*		*P<0.05*				*n/a*									
	Crop x year	*n/a*		*P<0.05*				*n/a*			*P<0.001*		*P<0.001*				
EMEL	Grass	11	3	204	200	158	0		386	283			238	165	148	118	20.2
	Maize	5	6	64	49	221	125	80	262	110	34.5	31.7	166	152	97	-42	143.8
	SBM	19	6	133	134	59	0	120	186	164	7.1	7.3	127	131	59	33	43.4
	WBF	0	0	*	*	*	*	200	*	*	*	*	*	*	*	*	*
	WO	2	0	139	*	0	*	130	169	*	4.5	*	92	*	77	*	*
	WOSR	6	3	194	211	0	120	220	224	271	3.7	3.8	112	115	112	156	-39.8
	WWF	16	3	223	168	81	0	240	272	198	8.9	8.9	176	181	96	17	82.4
	WWM	2	0	206	*	0	*	240	236	*	9.0	*	171	*	65	*	*
	Crop	*n/a*		*P<0.001*		*P<0.001*		*n/a*	*P<0.001*		*P<0.001*		*P<0.001*		*P<0.01*		
	Year	*n/a*						*n/a*	*P<0.05*						*P<0.05*		
	Crop xyear	*n/a*				*P<0.001*		*n/a*									
MSA +EMEL	*Catchment*	*n/a*		*P<0.001*		*P<0.05*		*n/a*									
	Catch. x crop	*n/a*		*P<0.001*		*P<0.05*		*n/a*			*P<0.01*						
	Catch. x crop x yr	*n/a*						*n/a*			*P<0.001*						

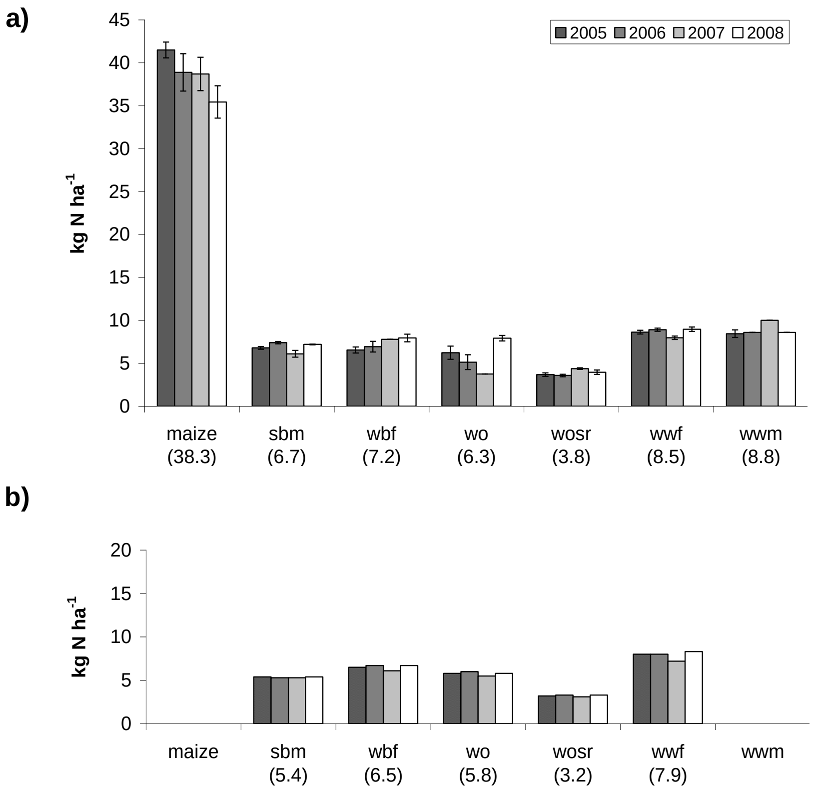


Figure 5-5: Average yield (t ha^{-1}) in (a) MSA/EMEL and (b) UK (BSFP, 2009) between 2005 and 2008. 2005-2008 averages shown in parentheses for comparison between UK and MSA / EMEL. No national data available for maize or WWM.

Effect of fertiliser recommendations on field budgets

With few farms adopting fertiliser recommendations only (see section 3.5.1) assessments of the effect of fertiliser recommendations on field budgets was confounded by responses to other mitigation methods. Significant reductions in fertiliser use were observed in MSA only, however this was accompanied by an increase in manure use. In EMEL fertiliser inputs tended to decrease but not significantly. Importantly though, fertiliser applications prior to mitigation were predominately lower than RB209 recommendations suggesting impact would have been minimal (Table 5-8). However recommended applications assume full accounting of manure N and should therefore be compared to fertiliser + manure inputs. On this basis inputs prior to mitigation were higher than recommended for maize, SBM, WO and WOSR, exposing opportunities for a reduction in fertiliser through improved manure accounting. However of these crops, only maize

(EMEL+MSA) and SBM (MSA only) saw a reduction in fertiliser inputs, and only for maize did this result in lower total inputs. For SBM the reduction in fertiliser was outweighed by an increase in manure. Post mitigation inputs highlight opportunities for (further) improvements relative to fertiliser recommendations for all crops except WWF (EMEL only).

Effect of manure management on field budgets

With the exception of WOSR, fields receiving manure had larger surpluses than those receiving fertiliser only (Figure 5-6). The difference was especially marked on maize where fields receiving manure displayed surpluses 164.86kg N ha^{-1} higher than those that did not. However it should be noted that only four maize fields did not receive manure. For crops typically receiving large quantities of manure (grass and maize) surplus reductions were much larger where manure was applied compared to those fields receiving fertiliser only (Figure 5-7). This highlights the greater potential for improved manure management compared to improved fertiliser use which as previously noted was generally in line with recommendations.

Fields were grouped according to the presence of a manure management plan (MMP), and farms not using manure excluded. Although differences before and after mitigation were not quite significant at the 95% confidence limit ($p=0.051$), crop and MMP were significant factors (Figure 5-8). Reductions in surpluses were larger on grass and maize fields, and surplus increases smaller on WOSR fields where MMP were followed; a significant crop x MMP interaction confirmed that responses to MMPs were crop specific. MMP also had a significant impact on fertiliser applications (crop $p<0.001$, year $p<0.05$, MMP $p<0.05$), and similar to surpluses, a significant interaction between crop and MMP response was observed. Reductions in fertiliser application were greater on grass, maize, and SBM fields where MMP were in place (Figure 5-9). However slightly larger reductions in fertiliser and surpluses were observed on non MMP WWF fields. Increases in fertiliser were observed on WOSR fields although the increase was smaller on MMP fields. Despite a reduction in fertiliser applications, the reduction in SBM surpluses was larger on non MMP fields. The effect of MMPs on manure inputs was not significant.

With limited change observed in manure management but significant differences observed in terms of fertiliser application and overall surpluses, MMP agreements appear to have had greater impact on manure N accounting than manure applications rates. This finding supports results across all farms where fertiliser inputs tended to decrease but manure inputs increase post mitigation.

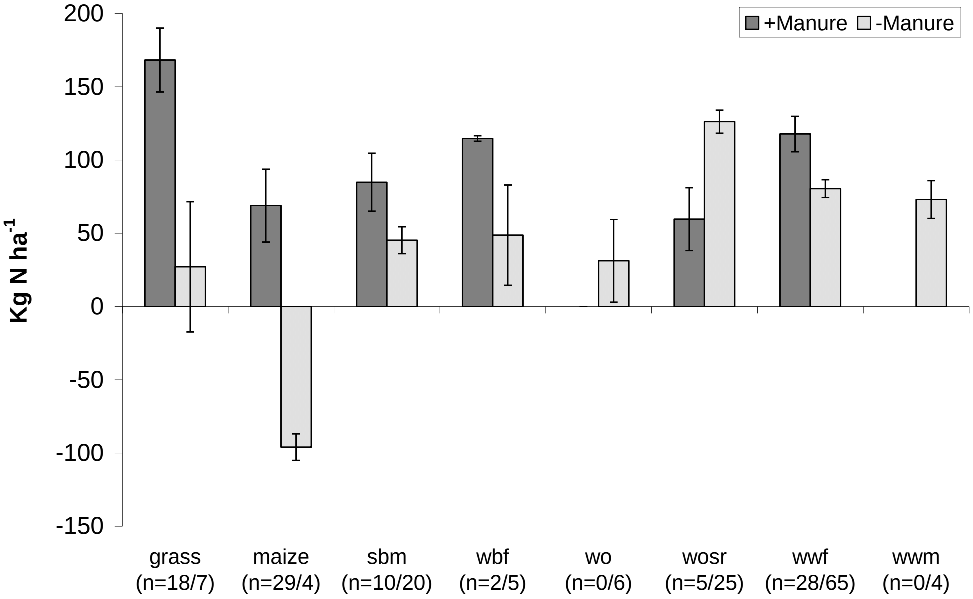


Figure 5-6: Soil surface surpluses (kg N ha^{-1}) for fields receiving and not receiving manure between 2005 and 2007. Mean and standard error shown with sample sizes before and after mitigation in parentheses. Differences between crops and +/- manure significant to p<0.001.

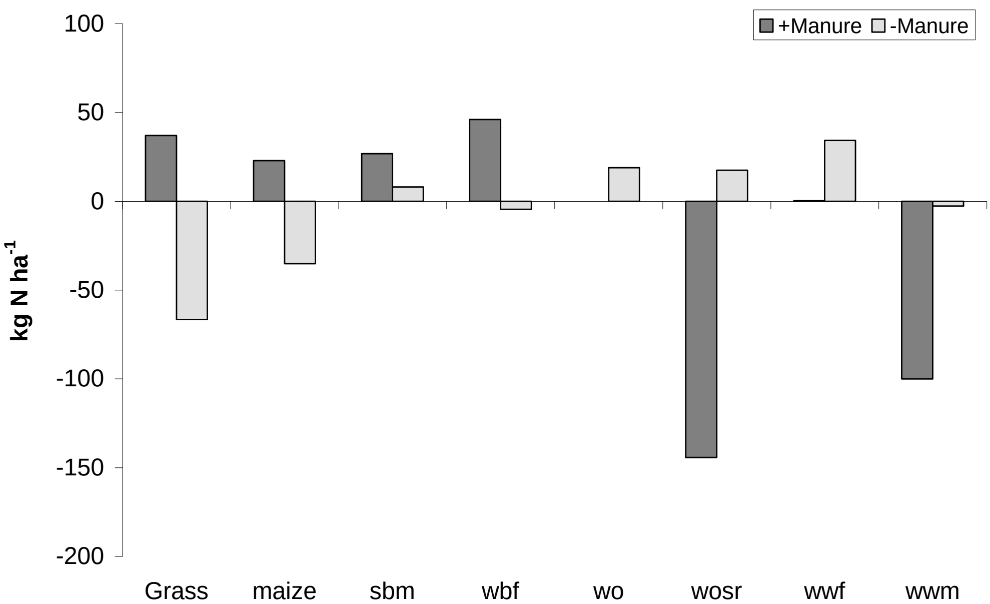


Figure 5-7: Change in average surplus (kg N ha^{-1}) on fields receiving / not receiving manure before and after mitigation.

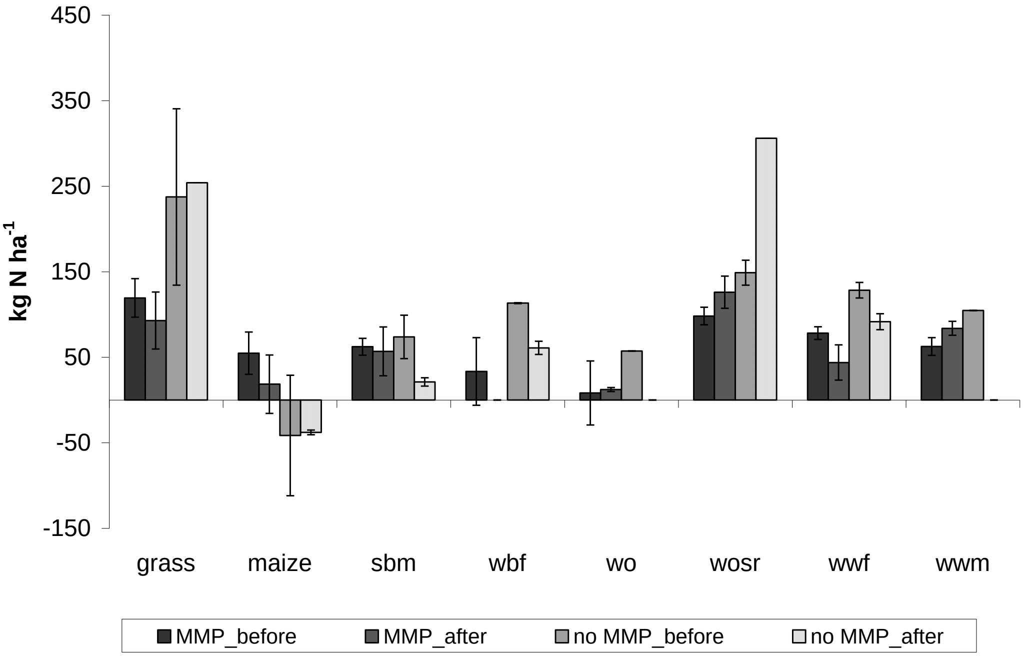


Figure 5-8: Comparison of soil surface surpluses (kg N ha^{-1}) before and after mitigation on fields with / without MMP agreements. Mean and standard error shown. Sample sizes not shown for clarity however 65% of fields were on farms adopting MMPs. Differences between crop, year and +/- MMP significant to $p<0.001$, $p=0.051$, $p<0.01$ respectively. Crop x MMP interaction significant to $p<0.05$.

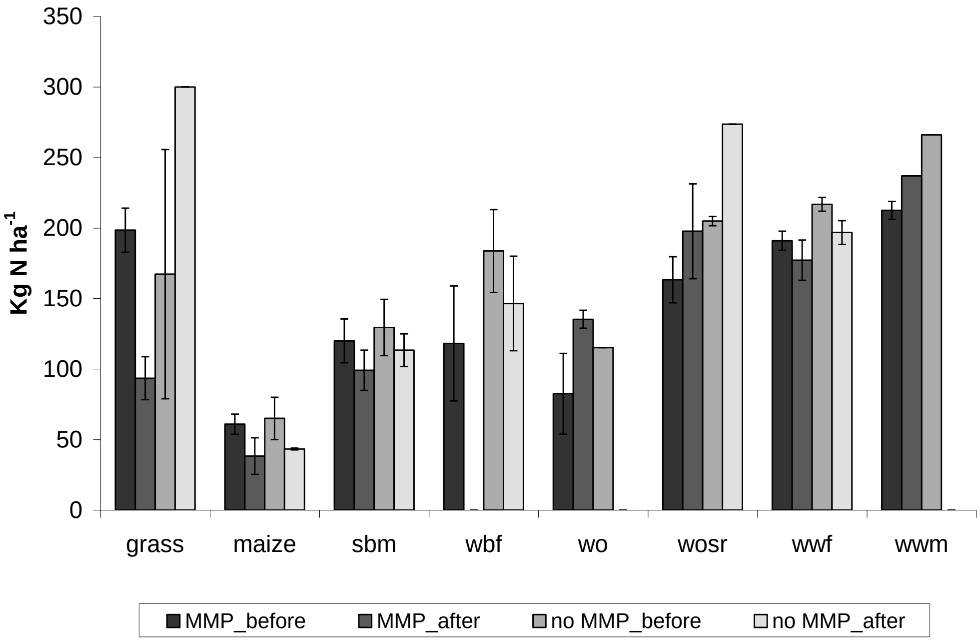


Figure 5-9: Comparison of fertiliser applications (kg N ha^{-1}) before and after mitigation to fields with / without MMP agreements. Error bars show standard error. Sample sizes not shown for clarity however 65% of fields were on farms adopting MMPs. Differences between crop, year and +/- MMP significant to $p<0.001$, $p=0.051$, $p<0.01$ respectively. Crop x MMP interaction significant to $p<0.05$.

On a farm – crop specific basis results concurred with those observed across all fields, with maximum impact observed on maize fields. The reduction in gross manure observed across all fields was evident at the farm scale, and in agreement with the fertiliser reduction associated with MMPs, improvements in manure accounting were also identified. However reductions were not restricted to between 2007 and 2008. For SBM and WWF, the impact of MMP on farm nutrient management was minimal, a reflection of good baseline manure management and greater reliance on fertiliser inputs. Farms not adopting MMPs were however more likely to make large unaccounted manure applications, suggesting the degree of existing agri-environmental awareness is reflected in the level of mitigation employed. In doing so, the potential for mitigation induced change is reduced.

Effect of cover crops on field budgets

The effect of cover crops on spring cropped fields (maize and SBM) was investigated by comparing pre-mitigation surpluses on bare fields with post-mitigation surpluses on both bare and cover cropped fields. A reduction in surpluses was observed post mitigation across all fields irrespective of over winter cover, however reductions were larger where cover crops had been grown (Figure 5-10). Differences between pre- and post-mitigation surpluses (both bare and cover cropped field) were however not significant. Wide variation in manure inputs resulted in wide variability especially for maize. On both a before vs. after and a 2008 only with vs. without basis, fertiliser and manure applications did not differ significantly between fields with / without a cover crop.

Farm specific responses support the insignificant response to cover crops with increased fertiliser applications often associated with their establishment. Although immobilised within the cover crop, increased inputs were not balanced unless the crop was grazed or cut. Lower 2008 surpluses on cover cropped fields may reflect a larger response to GAP and MMP on agri-environmentally aware farms. This further supports the idea that different ‘types’ of farms adopted different levels of mitigation.

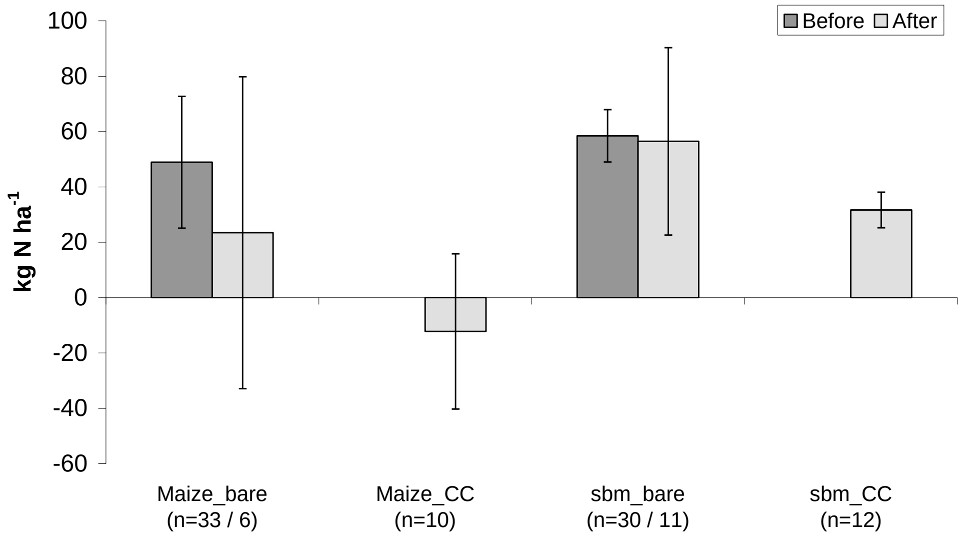


Figure 5-10: Effect of cover crop (CC) on maize and spring barley (SBM) soil surface surpluses (kg N ha^{-1}). Mean and standard error shown with sample sizes before and after mitigation in parentheses. Differences before and after mitigation not significant to $p<0.05$ for bare or cover cropped fields. Interactions between crop, year and +/- cc non significant.

Effect of spring manure applications on field budgets

Despite a number of farms agreeing to delay autumn manure applications to the spring (EGAP3 – see section 3.5.1), only one farm made significant changes to manure application dates. Based on a single farm's response, the apparent impact of this measure may not be representative of other farms, and results should be treated with caution.

Farm data highlighted a substantial change in manure application dates to WWF, delayed from September / October in 2005-2007 to late March in 2008. Of the crop surpluses obtained (grass, SBM, WOSR, and WWF), surpluses decreased post mitigation on grass and WWF fields, although there was considerable variability in grass surpluses (Figure 5-11). Average WWF surpluses fell 34.4 kg N ha^{-1} to -33.0 kg N ha^{-1}. Significantly less manure was applied to WWF, WOSR and SBM in 2008, the largest reduction observed on WWF fields. However only on WWF fields did this reduction translate to a lower surplus, with increased fertiliser applications compensating for lower manure N on WOSR and SBM fields. Although drawn from a limited dataset the impact of 'delayed manure applications' differed to that of MMPs with responses transpiring in manure application rates directly and not through reduced fertiliser inputs.

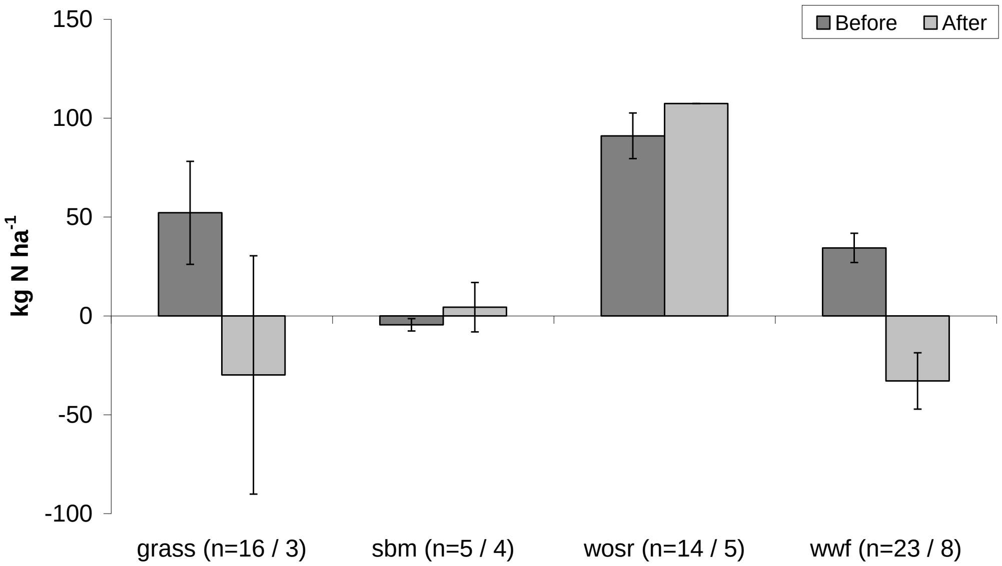


Figure 5-11: Effect of spring manure application on Bagber soil surface surpluses. Mean and standard error shown with sample sizes before and after mitigation in parentheses. Prior to mitigation fields received manure late autumn / early winter. During the mitigation year, WWF fields were subject to spring manure applications in the 'mitigation' year whilst all other crops continued to receive manure in late autumn / early winter. Differences between crops significant to p<0.001, and years to p<0.05. Interaction between crop and year close to accepted p<0.05 level of significance at 0.075.

5.4 Further development of field scale budget approaches

5.4.1 Mitigation scenarios

5.4.1.1 Introduction and methodology

Limited resources and time meant the range of mitigation methods available to farmers during the WAgriCo project was narrow, and the degree of change induced by them relatively small. In an attempt to investigate the sensitivity of budgets to a wider range of more influential methods, a range of mitigation scenarios were developed and simulations performed.

Mitigation methods applicable at the field scale were identified, and their likely impact on field scale budgets considered (Table 5-9). Inputs / outputs affected by each method were identified and those affected by a number of mitigation options developed as a mitigation scenario (Table 5-10). In doing so, each scenario simulates the implementation of a range of mitigation options. Where methods required the alteration of coefficients (e.g. manure N), sensitivity analysis was highlighted as a suitable method of investigation. Details of sensitivity analyses can be found in section 5.4.2.

MSA / EMEL farm data prior to mitigation (2005-2007 average) was adjusted to simulate the impact of the five mitigation scenarios. Mitigation scenario surpluses were calculated on a crop average basis before being compared to 2005-2007 averages (refered to in sections 5.4 and 6.4 as 'baseline' data) to assess impact and effectiveness. Results were also upscaled to assess potential impact at the catchment scale; methodological details and results can be found in section 5.4.3.2.

5.4.1.2 Assumptions and justifications

1. 'Baseline' nutrient utilisation was considered sub optimal; fertiliser applications typically exceed crop offtake by 25kg N ha^{-1} (Goulding et al., 2000) and N supplied via manure is not always fully accounted for in fertiliser planning (Chambers et al., 2000). It is therefore assumed that moderate reductions in inputs would not compromise yield. This is in agreement with Oenema et al., (2001) who expected crop yields to not / hardly decrease despite a reduction in fertiliser, as a result of improved utilisation of manures. Moreover, no compensatory behaviour is assumed to take place. Where fertiliser is reduced, manure and feed inputs for example cannot be modified. Similarly, an increase in nutrient availability will not increase yield. Nutrients applied above recommended rates have minimal impact on yield (Goulding et al., 2000). The sensitivity of balance calculation to yield is however considered in section 5.4.2.
2. Fertiliser reductions of up to 100% have been reported in field scale measurement and budget studies (e.g. Sieling and Kage, 2006). However, given that yields are assumed to remain constant, and fields contribute to commercial not research farms where profitability is at the forefront of decision making, smaller reductions would be more appropriate. Oenema et al., (2001) reports the impact (in part) of a 17% reduction in fertiliser on the farm surpluses of commercial dairy farms. With WAgriCo fertiliser applications typically about 10%

higher than national averages, a 20% reduction was selected on the grounds that modified applications would be less than national averages yet be in agreement with other studies where impact on yield was expected to be minimal.

3. Accounting for manure on a total N (TN) basis reflects a maximum impact scenario. Total manure N was subtracted from fertiliser applications after ammonia volatilisation losses. The assumption that manure N is not already accounted for holds true for most grass fields and for a large number of arable fields. Where manure N has already been accounted, limiting fertiliser reductions to 0kg N ha^{-1} will limit double counting.
4. Fertiliser recommendation systems such as PLANET (ADAS, 2008) and RB209 (Defra, 2000) only include readily available N (RAN) when accounting for manure N, reflecting the unavailability of organic N prior to mineralisation. For consistency and simplicity, NH3 losses were assumed to be proportionally the same from TN and RAN, with ammonia loss from RAN calculated in the same way as for TN using an accountability factor. However RAN will by definition be more at risk of volatilisation, and so where incorporation delays are long, net gain to the soil may be less than stated.
5. Incorporation delays were selected on the grounds that FYM cannot be incorporated on grassland, and that deep injection is the only means of incorporating slurry on grassland. A 24 hour delay was chosen for arable fields on the grounds that this was realistic but would represent a significant improvement on most farms. 24% of WAgriCo farms routinely incorporate manures within 24 hours. Nationally 26% of farm yard manure (FYM), 25% cattle slurry and 56% poultry manure is incorporated within 24 hours (BSFP, 2009).
6. A 20% reduction in gross manure was chosen to ensure consistency and facilitate comparison with fertiliser recommendation impact.

Table 5-9: Field scale mitigation methods – exploring budget sensitivity and mitigation scenario development.

Category	Mitigation method	Effect on budget	Mitigation scenario (see Table 5-10)
Land use	Convert arable land to extensive grassland	Extensive grass field budgets	n/a
Soil management	Establish cover crops in the autumn	Field budgets with cover crop[1]	n/a
	Establish in-field grass buffer strips	Reduction of inputs / outputs[1]	n/a
Livestock management	Reduce overall stocking rates on livestock farms	Reduce manure applications / reduce grazing and grazing excreta[1]	Scenario 5
	Reduce the length of the grazing day or grazing season	Adjust grazing coefficients[1,2]	Sensitivity analysis

Category	Mitigation method	Effect on budget	Mitigation scenario (see Table 5-10)
	Reduce dietary N and P intakes	Alter cropping (increase maize). Reduce excreta N	Sensitivity analysis
	Adopt phase feeding of livestock	Adjust excreta N	Sensitivity analysis
Fertiliser management	Use a fertiliser recommendation system	Reduce fertiliser application[3]	Scenario 1
	Integrate fertiliser and manure nutrient supply	Reduce fertiliser application, reduce manure	Scenarios 2 and 3
	Reduce fertiliser application rates	Reduce fertiliser application	Scenario 1
Manure management	Increase the capacity of farm manure (slurry) stores	Adjust manure N content[2]	Sensitivity analysis
	Minimise the volume of dirty water produced	Reduce manure application / reduced N content[2]	Sensitivity analysis/ Scenario 5
	Adopt batch storage of solid manure	Reduce N content	Sensitivity analysis
	Compost solid manure	Reduce N content	Sensitivity analysis
	Change from slurry to a solid manure handling system	Adjust N content of manure[2]	Sensitivity analysis
	Incorporate manure into the soil	Alter accountability factor, manure application rate	Scenario 4
	Transport manure to neighbouring farms	Reduce manure application	Scenario 5
	Incinerate poultry litter	Reduce manure application	Scenario 5

[1] Budgets do not reflect improved resistance to nutrient transport.

[2] Budgets do not fully reflect changes in the timing of nutrient applications. The higher risk of loss associated with nutrient applications in the autumn / winter and before heavy rain is poorly reflected in budget results.

[3] Assuming fertiliser recommendations are lower than current application rates. However this was not the case within the WAgriCo catchments.

Table 5-10: Description of mitigation scenarios and simulation approach

Mitigation Scenario		Approach	Relevant assumptions / justifications
1	Reduced fertiliser	20% reduction in gross fertiliser inputs	1, 2
2	Integrated fertiliser and manure (TN method)	Fertiliser input reduced by net TN to a minimum of 0N. Assumes no prior accounting of manure N.	1, 3
3	Integrated fertiliser and manure (RAN method)	Fertiliser input reduced by net RAN to a minimum of 0N. Assumes no prior accounting of manure N and that NH3 losses from RAN are proportionally the same as from TN.	1, 4
4	Rapid incorporation of manure	Manure to arable crops inc. within 24hrs. Slurry applied to grass via deep injection. FYM to grass remains unincorporated.	1, 5
5	Reduced gross manure	20% reduction in gross manure inputs.	1, 6

5.4.1.3 Mitigation scenario results

Modelled responses to mitigation scenarios are summarised in Table 5-11. Surplus reductions were observed following the simulation of scenarios 1, 2, 3 and 5 (Figure 5-12). Scenario responses were crop specific, reflecting 'baseline' surpluses, fertiliser inputs, and manure management. With most fields receiving fertiliser, scenario 1 was most applicable and had the largest overall impact. Absolute improvements were largest where fertiliser inputs were highest (e.g. WOSR) and where reductions in fertiliser were large relative to 'baseline' surpluses (e.g. WO). Scenarios 2-5 targeted manure applications and were only applicable to fields receiving manure; they had no effect on WO or WWM. Correspondingly, those fields receiving the largest manure applications (grass and maize) were most affected. Large 'baseline' surpluses meant large absolute reductions on grass fields did not however translate to large relative reductions. By accounting for total manure N, scenario 2 was more effective than scenario 3 and on maize fields resulted in the largest absolute and relative improvements of any crop – mitigation combination (60.39kg N ha^{-1} / 97.16%). However, it should be noted that accounting for TN may overestimate the amount of crop available N and jeopardise profitable yields. Despite the same 20% reduction, fertiliser reductions were more effective than reductions in manure. Only on maize fields were manure inputs larger than fertiliser inputs and scenario 5 more effective than scenario 1.

Table 5-11: Summary of mitigation scenario impact on kg N ha^{-1} and % change from 'baseline' result basis

Scenario	kg N ha^{-1} basis	% change from 'baseline' basis
1	Surpluses decreased by 12.4 -45.2 kg N ha^{-1} Maximum impact on WWM due to high fertiliser inputs. Most effective scenario on all crops except grass, maize and WBF.	Surpluses decreased by 65.1-25.3% Maximum reductions on WO where fertiliser inputs are high compared with surpluses. Scenario yielded consistently good results.
	Applicable to most fields. Improvements in surpluses typically exceeded fertiliser reduction.	
2	Surpluses decreased by 0-60.4 kg N ha^{-1}. Maximum impact was observed on grass fields due to high manure and fertiliser inputs (maximum fertiliser reduction potential). WO and WWM were unaffected because they received no manure. Second most effective mitigation method.	Surpluses decreased by up to 97.7% (maize – large reduction relative to surplus), the largest improvement seen on any crop / mitigation combination. Less effective where manure inputs were low.
	The method is not applicable to all fields and may over estimate the amount of N available to crops	

Scenario	kg N ha^{-1} basis	% change from 'baseline' basis
3	Reductions ranged from 17.4 (maize) to 0 (WWM) kg N ha^{-1}. Reductions were largest on maize, grass and WBF fields (consistent with scenario 2).	Surplus reductions of 0 – 30.6% were observed Maximum relative change on maize and WBF. Grass effectiveness appeared relatively low due to high 'baseline' surpluses.
	A more realistic representation but with modest results compared with scenario 2. Results are similar to a 20% reduction in gross manure.	
4	Maximum increases were observed in grass (6.1 kg N ha^{-1}) and maize (6.9 kg N ha^{-1}). The method was also effective on grass.	A maximum increase of 14.00% was observed on maize fields.
	Surpluses increased because rapid incorporate conserves N increasing net manure input. Impact was least on arable crops (excluding maize) which tended to receive less manure and already followed good practice in terms of rapid incorporation. The magnitude of impact was small compared with other scenarios.	
5	Reductions in surpluses ranged from 29.5 to 0 kg N ha^{-1}. Maximum reductions were observed on maize fields, a result of large manure applications.	Reductions ranged from 0-60.3%. Largest reductions on maize fields where manure inputs were high.
	Impact was comparable to scenario 3. No impact on WO or WWM as scenarios 2-4.	

By reducing volatilisation losses, rapid incorporation (scenario 4) increased net manure inputs and resulted in larger post mitigation surpluses (Figure 5-12). Magnitude of increases reflected the size of manure applications and potential to reduce application- incorporation delay. Largest increases were therefore observed on grass and maize fields where application rates and incorporation delays were both high. Larger post mitigation surpluses highlight the risk of pollution swopping whereby a reduction in losses of one pollutant increases the loss of another. Nutrient conservation should be factored into nutrient management to ensure ammonia losses are not reduced at the expense of increased nitrate loss.

Crop specific responses complicate comparison of modelled scenario and observed WAgriCo mitigation impact. For maize, WO and WWF WAgriCo reductions are of a similar magnitude to the most effective mitigation scenario. For grass and SBM, WAgriCo reductions occupy an intermediate position with modelled responses to scenarios 1 and 2 exceeding observed improvements. However on WBF, WOSR and WWM fields, WAgriCo reductions were surpassed by all mitigation scenarios. Where modelled responses to mitigation exceed observed responses, scope for further improvement is exposed.

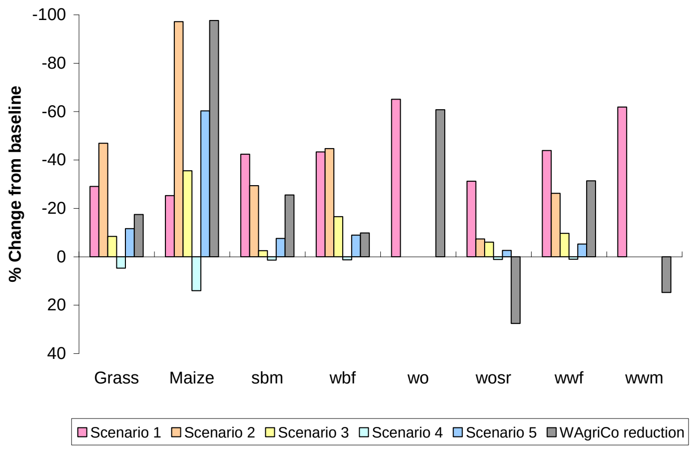


Figure 5-12: Modelled % change in surplus under mitigation scenarios compared to % change observed in MSA / EMEL following WAgriCo mitigation. % Change relative to 2005-2007 average.

5.4.2 Sensitivity analysis

5.4.2.1 Introduction

Sensitivity analysis, aimed at investigating the sensitivity of budget calculations to variables and coefficients was conducted to a) investigate the impact of possible uncertainty (in input data and coefficients) on surpluses, and b) to investigate additional mitigation situations simulated by coefficient adjustment.

To explore the impact of input data unreliability and coefficient variability, the relative uncertainty of input data (‘variables’) and fixed values (‘coefficients’) were compared (Table 5-12), and those representing higher levels of uncertainty selected for inclusion in sensitivity analysis. Manure data was identified as most unreliable with missing / approximate values justifying further investigation. Yield was selected on the grounds that 2008 yield figures were estimated on a number of farms due to late harvest, and because crop offtake, a function of yield, represents the only balance output. Although fertiliser application data was perhaps most reliable, large inputs mean small errors have a disproportionally large impact on surpluses, justifying the inclusion of fertiliser input in sensitivity analysis. With only grass fields grazed, and recently improved excreta N values, excreta and grazing variables were not directly included in the analysis. Grass N content and grazing rate were however included in coefficient investigations. The inclusion of manure, yield and fertiliser is in agreement with the sensitivity analysis conducted by Campling et al., (2005).

In terms of coefficients, manure N was selected due to sampling difficulties and heterogeneous N content, and for mitigation simulation purposes amongst which a reduction in manure N is prevalent. Accountability factors were included to address uncertainty attached to their recent development (see chapter 7). An abundance of crop N values in the literature and issues associated with their selection necessitated inclusion of crop N. On grassland grazing rate governs grazing offtake making it a fundamental calculation component and the subject of mitigation. As previously mentioned excreta N values were considered less uncertain following a recent review (Smith and Cottrill, 2007).

Table 5-12: Relative uncertainty of field budget input data and associated coefficients

	Variable	Uncertainty	Coefficient	Uncertainty
Inputs	Fertiliser	*Low*	n/a	
	Manure	*Medium/ high*	Manure N	*High*
			Acc. factor	*Medium*
	Excreta	*Medium*	Excreta N	*Low / medium*
	Atmos. depos.	*Low*	n/a	
Output	Crop offtake (yield)	*Medium*	Crop N	*Medium*
	Grazing offtake	*Medium*	Crop N	*Medium*
			Grazing rate	*Medium*

5.4.2.2 Methodology

Using a procedure similar to that employed by Campling et al., (2005), variables and coefficients were adjusted by increments of 10% to a maximum of +/- 20% and balances recalculated (all fields, all years). Minimum adjusted fertiliser applications and yield values were less than national averages (Table 5-13, Table 5-14); local fertiliser applications were approximately 10% higher than national averages, and yield 20% higher. Adjusted literature sourced coefficients extended beyond the limits of values obtained locally (Table 5-15 and Table 5-16). (Locally sourced values were not included directly due to limited and unrepresentative sampling). Modified accountability factors were compared to original values with respect to current practices to ensure adjusted values were feasible (Table 5-17). Adjusted grazing rates are shown in Table 5-18.

Table 5-13: Fertiliser application rates (kg N ha^{-1}) used in sensitivity analysis compared to average values in MSA / EMEL and UK. Sensitivity analysis inputs derived from average fertiliser application rates in MSA / EMEL between 2005 – 2008 adjusted +/- 10 and 20%.

Crop	Fertiliser application rate (kg N ha^{-1})								
	Sensitivity analysis values				MSA/EMEL 2005-2008 average				UK 2005-2008 average (BSFP, 2009)
	20%+	10%+	10%-	20%-	Ave.	SE	Max	Min	
Grass	196.9	180.5	147.7	131.3	164.1	15.2	337.3	0.0	101.50
Maize	64.4	59.1	48.3	43.0	53.7	5.7	123.0	0.0	63.50
SBM	140.4	128.7	105.3	93.6	117.0	7.0	228.3	0.0	106.75
WBF	175.6	160.9	131.7	117.0	146.3	21.9	214.0	0.0	146.25
WO	135.2	124.0	101.4	90.2	112.7	15.7	163.3	33.0	105.50
WOSR	219.7	201.4	164.8	146.5	183.1	10.3	273.6	41.0	195.75
WWF	239.0	219.1	179.3	159.4	199.2	4.3	294.5	86.8	181.50
WWM	276.7	253.7	207.5	184.5	230.6	7.9	266.0	204.5	214.25

Table 5-14: Yield values (t ha^{-1}) used in sensitivity analysis compared to average values in MSA / EMEL and UK. Sensitivity analysis values derived from average yield in MSA / EMEL between 2005 – 2008 adjusted +/- 10 and 20%.

Crop	Crop yield (t ha^{-1})								
	Sensitivity analysis values				MSA/EMEL 2005-2008 average				UK 2005-2008 average (Defra, 2009)
	20%+	10%+	10%-	20%-	Ave.	SE	Max	Min	
Grass	-	Na[1]	Na[1]	Na[1]	Na[1]	-	-	-	Nd[2]
Maize	39.8	33.2	33.2	33.2	33.2	36.5	29.9	26.6	Nd[2]
SBM	7.9	6.6	6.6	6.6	6.6	7.3	5.9	5.3	5.4
WBF	8.5	7.1	7.1	7.1	7.1	7.8	6.4	5.7	6.5
WO	8.3	6.9	6.9	6.9	6.9	7.6	6.2	5.5	5.8
WOSR	4.8	4.0	4.0	4.0	4.0	4.4	3.6	3.2	3.2
WWF	10.3	8.6	8.6	8.6	8.6	9.5	7.7	6.9	7.9
WWM	10.1	8.4	8.4	8.4	8.4	9.2	7.6	6.7	Nd[2]

[1]Due to variation in grass utilisation it was not possible to calculate an average grass yield.

[2] No data

Table 5-15: Manure N coefficients (kg N t^{-1}) used in sensitivity analysis compared to measured values in MSA / EMEL. Sensitivity analysis values derived from standard manure N contents in MANNER (Chambers et al., 1999) adjusted +/- 10 and 20%. Insufficient sampling in MSA / EMEL to derive sensitivity analysis inputs from MSA / EMEL values.

Manure type	Manure N content (kg N t^{-1})								
	Sensitivity analysis values					MSA/EMEL 2005-2008 average			
	Original	20%+	10%+	10%-	20%-	Ave.	SE	Max	Min
beef slurry	2.3	2.76	2.53	2.07	1.84	nd[1]	-	-	-
dairy slurry	3	3.60	3.30	2.70	2.40	nd[1]	-	-	-
dirty water	1.5	1.80	1.65	1.35	1.20	1.64	n/a	1.64	1.64
cattle FYM	6	7.20	6.60	5.40	4.80	5.05	0.49	7.57	3.58
layer manure	16	19.20	17.60	14.40	12.80	5.72	0.33	6.21	4.75
broiler litter	30	36.00	33.00	27.00	24.00	19.20	1.89	21.09	17.31

[1] nd = no data

Table 5-16: Crop N coefficients (kg N t fw^{-1}) used in sensitivity analysis compared to measured values in MSA / EMEL. Sensitivity analysis values derived from standard crop N contents (ADAS, 2008) adjusted +/- 10 and 20%. Insufficient sampling in MSA / EMEL to derive sensitivity analysis inputs from MSA / EMEL values.

Crop	Crop N content (kg N t fw^{-1})								
	Sensitivity analysis values					MSA/EMEL 2005-2008 average			
	Original	20%+	10%+	10%-	20%-	Ave.	SE	Max.	Min.
Grass	6.8	8.16	7.48	6.12	5.44	nd[1]	-	-	-
Maize	4.8	5.76	5.28	4.32	3.84	nd[1]	-	-	-
Rye[2]	17	-	-	-	-	15.13	0.86	16.80	13.90
peas[2]	35	-	-	-	-	40.20	0.70	40.90	39.50
sbf[2]	17	-	-	-	-	17.27	0.85	22.80	13.90
sosr[2]	33	-	-	-	-	30.90	-	-	-
swf[2]	17	-	-	-	-	21.60	-	-	-
SBM	14	16.8	15.4	12.6	11.2	14.94	0.33	18.60	12.10
WBF	17	20.4	18.7	15.3	13.6	16.76	0.83	20.50	13.70
WO	17	20.4	18.7	15.3	13.6	16.06	0.54	23.10	13.80
WOSR	30	36	33	27	24	34.64	1.70	41.20	31.30
WWF	17	20.4	18.7	15.3	13.6	19.53	0.23	25.20	13.30
WWM	19	22.8	20.9	17.1	15.2	19.02	0.25	23.30	16.60

[1] nd = no data

[2]Sensitivity analysis restricted to ‘dominant crops’

Table 5-17: Accountability factors (derived from manure type and delay between application and incorporation) used in sensitivity analysis compared to observed incorporation delays in MSA/EMEL. Derivation of accountability factors discussed in chapter 7. Accountability factors adjusted +/- 10 and 20% for sensitivity analysis

Manure type	Incorporation delay	Current situation – no. fields [a]				Accountability factors				
		2005	2006	2007	2008	Orig.	20%+	10%+	10%-	20%-
Poultry	Not inc	-	-	3	3	0.43	0.52	0.47	0.39	0.34
	6-10 days	6	6	1	5	0.47	0.56	0.52	0.42	0.38
	24hrs	2	2	7	3	0.51	0.61	0.56	0.46	0.41
	6hrs	-	-	-	-	0.52	0.62	0.57	0.47	0.42
	Deep injection [b]	-	-	-	-	0.52	0.62	0.57	0.47	0.42
Cattle	Not inc	17	18	15	11	0.73	0.88	0.80	0.66	0.58
	6-10 days	3	4	2	6	0.74	0.89	0.81	0.67	0.59
	24hrs	2	4	5	2	0.77	0.92	0.85	0.69	0.62
	6hrs	-	-	1	-	0.82	0.98	0.90	0.74	0.66
	Deep injection [b]	-	-	-	-	0.86	1.032 [c]	0.95	0.77	0.69

[a] Number of fields within MSA/EMEL field budget dataset

[b] Slurry only

[c] Value replaced with 1 in calculations as > 100% retention not possible.

Table 5-18: Grazing rates (kg DM LU^{-1} day^{-1}) used in sensitivity analysis. Standard grazing rate (Hopkins, 2000) adjusted +/- 10 and 20% for sensitivity analysis.

Livestock	Grazing rates (kg DM LU^{-1} day^{-1})				
	Original	20%+	10%+	10%-	20%-
Dairy cattle	15	18	16.5	13.5	12
Beef cattle	12	14.4	13.2	10.8	9.6
Sheep	2	2.4	2.2	1.8	1.6

5.4.2.3 Results

Sensitivity to variables

Surpluses increased with increasing fertiliser and manure inputs although responses were crop and variable specific (Figure 5-13 a and b). Largest increases were observed where 'baseline' inputs were already high; grass, WOSR, WWF, WWM for fertiliser, and grass and maize for manure. On a % change : % change basis, results reflected the degree of change relative to surpluses. Maximum fertiliser sensitivity was observed for WO where the change in inputs was large relative to surpluses. With maize surpluses also low, change following manure adjustment also translated to high relative change. In contrast, high WOSR surpluses meant the ratio of change was small despite a large absolute change under adjusted fertiliser inputs. Fewer crops were sensitive to changes in manure inputs, reflecting the crop specific nature of applications (Figure 5-13b).

Responses to yield adjustment were the inverse of fertiliser and manure, with an increase in yield resulting in surplus reductions (Figure 5-13c). Maximum increases on a kg N ha^{-1} basis were observed where yields were high (maize, WWF, WWM), however low surpluses meant that on a relative basis sensitivity was also high on maize and WO. Maize, SBM, WBF and WO were most sensitive to yield, whilst high fertiliser applications meant WOSR, WWF and WWM were most sensitive to fertiliser (Figure 5-14).

Sensitivity to coefficients

Increasing manure N coefficients increased surpluses especially on fields receiving large amounts of manure such as maize (Figure 5-15a). Sensitivity was very low / negligible on crops receiving little / no manure such as WO and WWM. Following the removal of fields not receiving manure, the sensitivity of arable crops increased considerably especially on WBF, highlighting the ‘dilution effect’ of fields receiving no manure in the first analysis (results not shown). Increasing crop N coefficients led to a reduction in surpluses, the largest observed on grass due to high offtake via grazing and cutting (Figure 5-15b). Due to low surpluses and high / moderately high yields, maize and WO were most sensitive on a % change basis. SBM and WOSR were least sensitive to crop N due to lower yields. Sensitivity to accountability was similar to that of manure N, a result of both coefficients being a factor of the initial manure application. Accordingly maize was most sensitive to manure N and WO / WWM least sensitive (Figure 5-15c). Low initial surpluses meant sensitivity was also high for maize on a % change basis with relative impact almost 5 times that induced by accountability factors. For grass, sensitivity to grazing rate was identical to that of crop N, reflecting comparable representations in grass offtake calculations (data not shown).

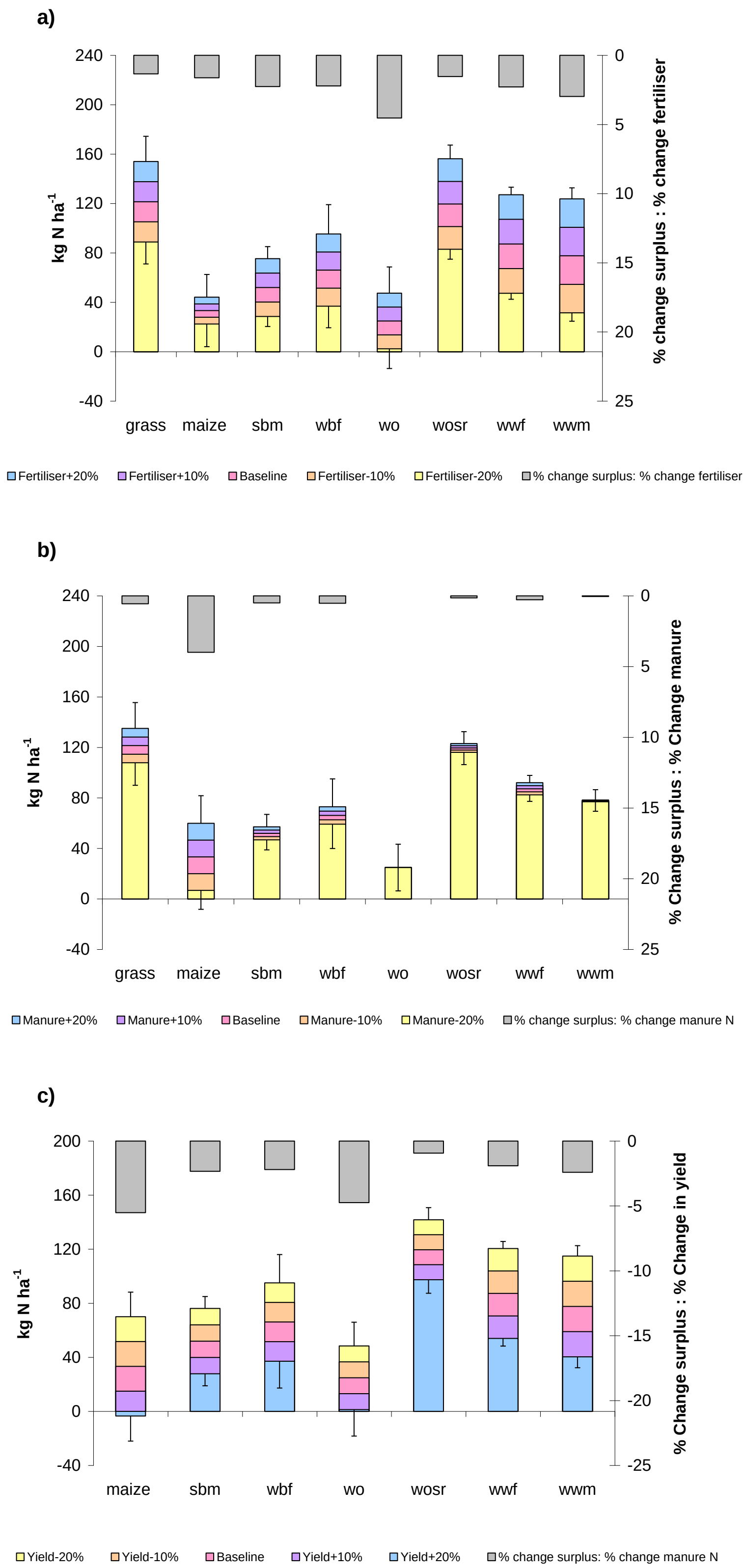


Figure 5-13: Sensitivity of soil surface balance to (a) fertiliser, (b) manure, (c) yield. Graphs show both surpluses (kg N ha^{-1}) and the ratio of % change between input and surplus.

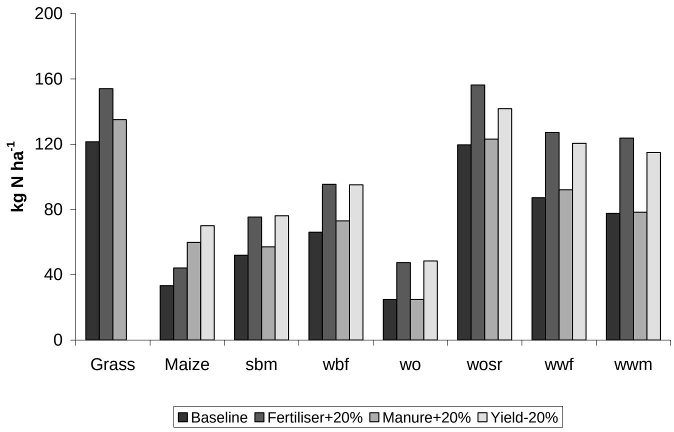


Figure 5-14: Sensitivity of soil surface surpluses to changes in fertiliser, manure and yield (kg N ha^{-1}). Grass yield sensitivity excluded due to grazing.

Of the coefficients investigated, sensitivity to crop N was much greater than manure N / accountability (Figure 5-16). This is because crop N affects crop offtake, the only balance output. Changes to crop N content translate directly to total output, resulting in surplus changes of up to 110% following a 20% reduction in crop N. Accountability and manure N in contrast affect just one of a number of inputs, and in some cases (0 manure N fields) have no impact on surpluses. With both accountability and manure N scaling manure inputs by 20%, sensitivity was identical. Where 0 manure N fields were removed, sensitivity to manure N and accountability increased on all crops except maize. Surpluses on non manure maize fields (n=9) were very low meaning the increase in average surpluses exceeded the increase in sensitivity to manure change (results not shown).

With sensitivity to crop N especially high, budgets were also re-calculated using maximum and minimum locally measured crop N values to investigate error induced by standard values from the literature. Although the difference between maximum and minimum crop N surpluses was high, standard crop N surpluses remain within the range of those calculated using locally measured values with the exception of WOSR (Figure 5-17). In the absence of a representative set of measured values these results support the use of standard values but suggest results may not provide an accurate field specific result.

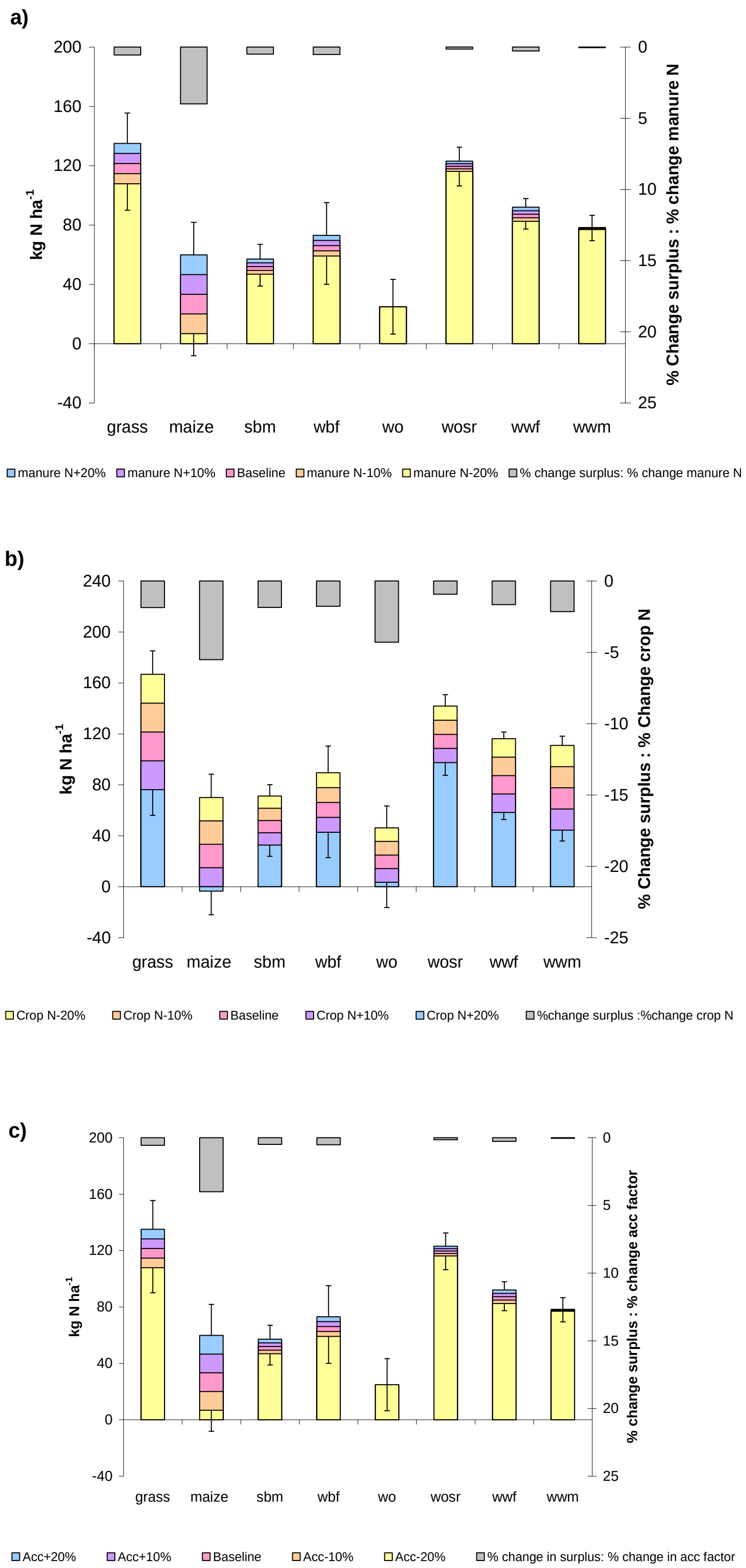


Figure 5-15: Sensitivity to (a) manure N, (b) crop N, and (c) accountability factor. Graph shows both surpluses (kg N ha^{-1}) and the ratio of % change between input and surplus.

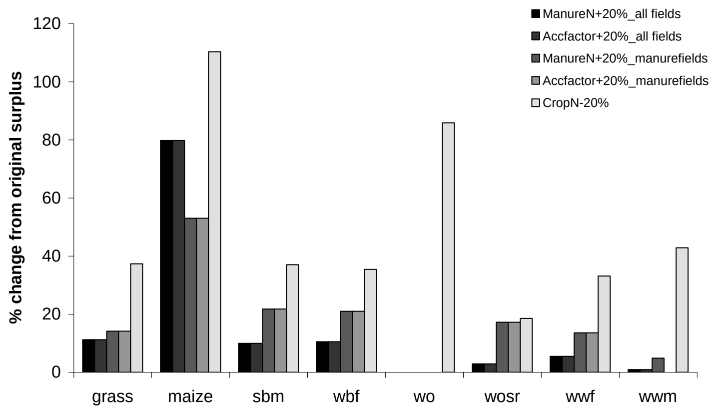


Figure 5-16: Comparison of the sensitivity of soil surface balance to manure N, accountability and crop N. Graph shows % change between 2005-2007 average and mean modified surplus.

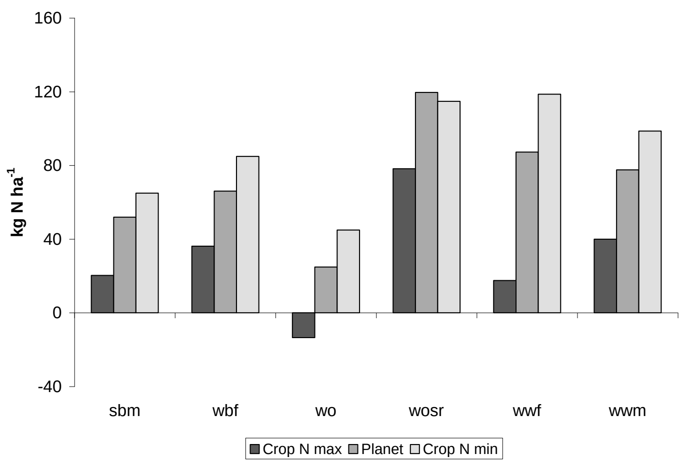


Figure 5-17: Comparison of soil surface surpluses calculated using maximum and minimum measured crop N values in MSA / EMEL between 2005 and 2008 with those obtained using crop N values from the existing PLANET farmgate budget methodology (Defra, 2005).

Variable vs. coefficient sensitivity

Sensitivity to variables and coefficients were crop specific, reflecting differences in nutrient management and yield (Figure 5-18). Factors affecting output induced greater variability in surpluses than those affecting inputs. Sensitivity to yield was higher than that of crop N due to its impact on straw offtake which was particularly relevant on winter oats. Reliance on mineral fertiliser meant sensitivity to fertiliser inputs was also high, exceeding that of yield on WOSR, WWF, and WWM due to high fertiliser inputs. The opposite was observed on maize, SBM, WBF and WO where sensitivity to yield exceeded that of fertiliser due to high offtake relative to surpluses (and low fertiliser in the case of maize). With all three manure factors adjusting manure inputs by 20%, responses were identical. Across all factors sensitivity was highest on maize and WO due to high manure inputs / high outputs and low surpluses. Across all crops, the degree of mitigation induced change was consistently exceeded by sensitivity to at least two factors, most commonly fertiliser and yield. In other words uncertainties in input data / coefficients have the potential to conceal apparent responses to mitigation. Accurate and complete input data and locally sourced coefficients are therefore essential if field scale budgets are to provide reliable assessments of mitigation effectiveness.

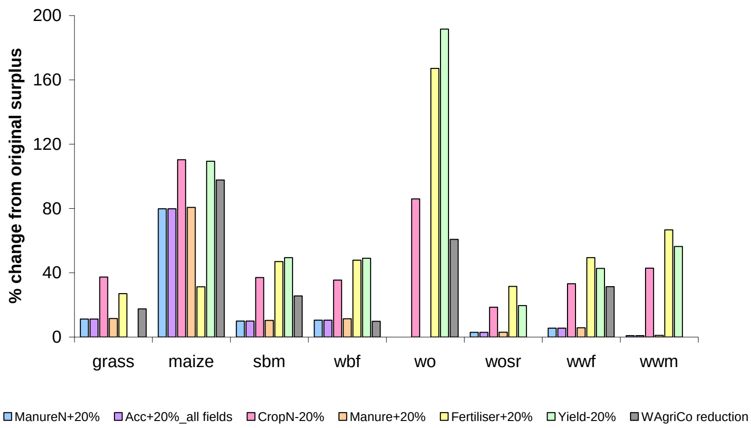


Figure 5-18: Relative sensitivity of soil surface balance of variables (fertiliser/ manure / yield) and coefficient (manure N / accountability / crop N sensitivity). Graph shows average % change between modelled surplus and 2005-2007 observed average.

5.4.3 Upscaling field surpluses to the catchment scale

Soil surface budgets have been used to assess nutrient management and mitigation effectiveness at the field scale. While existing results are useful for understanding mitigation impact and field specific nutrient management, it is not possible using only a subset of fields to comment on catchment impact or suggest how mitigation might translate to improvements in waterbody status. With both catchments predominately agricultural, investigating catchment wide trends could be obtained by calculating soil surface balances for all fields; the area weighted average of these fields representing an average catchment surplus. However as previously noted, calculating budgets for over 500 fields would be very time consuming. Therefore in order to address objective 4, an alternative means of calculating a catchment wide surplus based on the projection of existing field scale results was developed.

5.4.3.1 Calculating a catchment surplus

With maximum field scale variability attributed to crop type, crop average surpluses were projected across the each catchment based on cropping patterns. Average crop surpluses were calculated for the 8 dominant crops, and the proportion of each catchment occupied by these crops tabulated for 2005 - 2008. The fraction of each catchment occupied by each crop was then multiplied by the corresponding average surplus. The sum of these values provided an annual catchment average surplus (see Table 5-19). Approximately 40% of both catchments are grass, however due to grass quality / slope etc around 40% of this is unfertilised. Grass fields included in the soil surface budget subset belonged predominately to the fertiliser fraction, meaning an average unfertilised grass surplus was not available. With minimal grazing and cutting and zero farmer inputs, a 0 kg N ha^{-1} surplus for unfertilised grass has been adopted.

Table 5-19: Upscaling field scale results to the catchment scale – worked example for MSA in 2005.

Crop	Crop coverage (% of total catchment)	Average soil surface surplus (kg N ha^{-1})	Area weighted average soil surface surplus (kg N ha^{-1}) = % coverage / 100 x surplus
grass (fert.)	26.16	97.42	27.19
grass (unfert.)	15.40	0.00	0.00
maize	5.21	63.30	3.51
SBM	4.12	77.59	3.41
WBF	3.32	113.61	4.02
WO	3.25	-56.00	-1.94
WOSR	11.41	141.74	17.25
WW	24.89	86.47	22.95
Total	93.75		**76.39**

Concerns regarding the representivity and size of the field scale budget subset meant a number of different average crop surplus options were explored. The benefit of heightened sensitivity to spatial and temporal variability was compared to a reduction in input fields which may compromise representivity. Catchment surpluses were first derived from crop average surpluses from all 4 years, projected onto crop areas averaged across the same years (M1 and M2). Under M2 crop surpluses were catchment specific. Annual catchment surpluses were calculated using year specific crop surpluses, projected onto corresponding annual cropping

(M3). Only in MSA did the sample size permit doing this on a catchment specific basis.

Catchment projection results

Table 5-20: Catchment projection of field scale results (kg N ha^{-1})

Method	MSA					EMEL				
	2005	2006	2007	2008	Ave.	2005	2006	2007	2008	Ave.
M1	-	-	-	-	76.87	-	-	-	-	77.77
M2	-	-	-	-	72.92	-	-	-	-	84.78
M3	76.80	93.47	82.41	63.05	-	86.34	95.33	75.70	61.67	-
	'baseline' = 84.23			63.05	-	'baseline' = 85.79			61.67	-
M4	76.39	100.65	68.34	65.65	-	-	-	-	-	-
	'baseline' = 81.79			65.65	-	-	-	-	-	-

Using method 1 (M1), MSA and EMEL results were very similar, with slight differences reflecting variation in catchment cropping (Table 5-20). Where results were calculated on a catchment specific basis (M2), the EMEL catchment surplus was larger than MSA, a result of higher grass surpluses in EMEL than MSA. Methods 3 and 4 (M3 and M4) exposed changes in both cropping and nutrient management. In both catchments and under both methods, maximum surpluses were observed in 2006, a result of large grass surpluses that year, and minimum surpluses in 2008, a result of low grass and WW surpluses and a higher proportion of SBM fields with low surpluses. 2008 surpluses were also consistently lower than 'baseline' results. Results of methods 3 and 4 (M3 and M4) were comparable in all years except 2007 when a c.50kg N ha^{-1} reduction in the grass surplus (average of MSA+EMEL fields vs. MSA fields only) resulted in 14kg N ha^{-1} lower surplus under M4.

Results of MSA M4 projections are mapped in Figure 5-19. High grass surpluses explain the numerous high surplus fields in 2006. Although relatively high surpluses were observed on a number of fields in 2008, a large number of low surplus fields meant the catchment average was lower than all previous years.

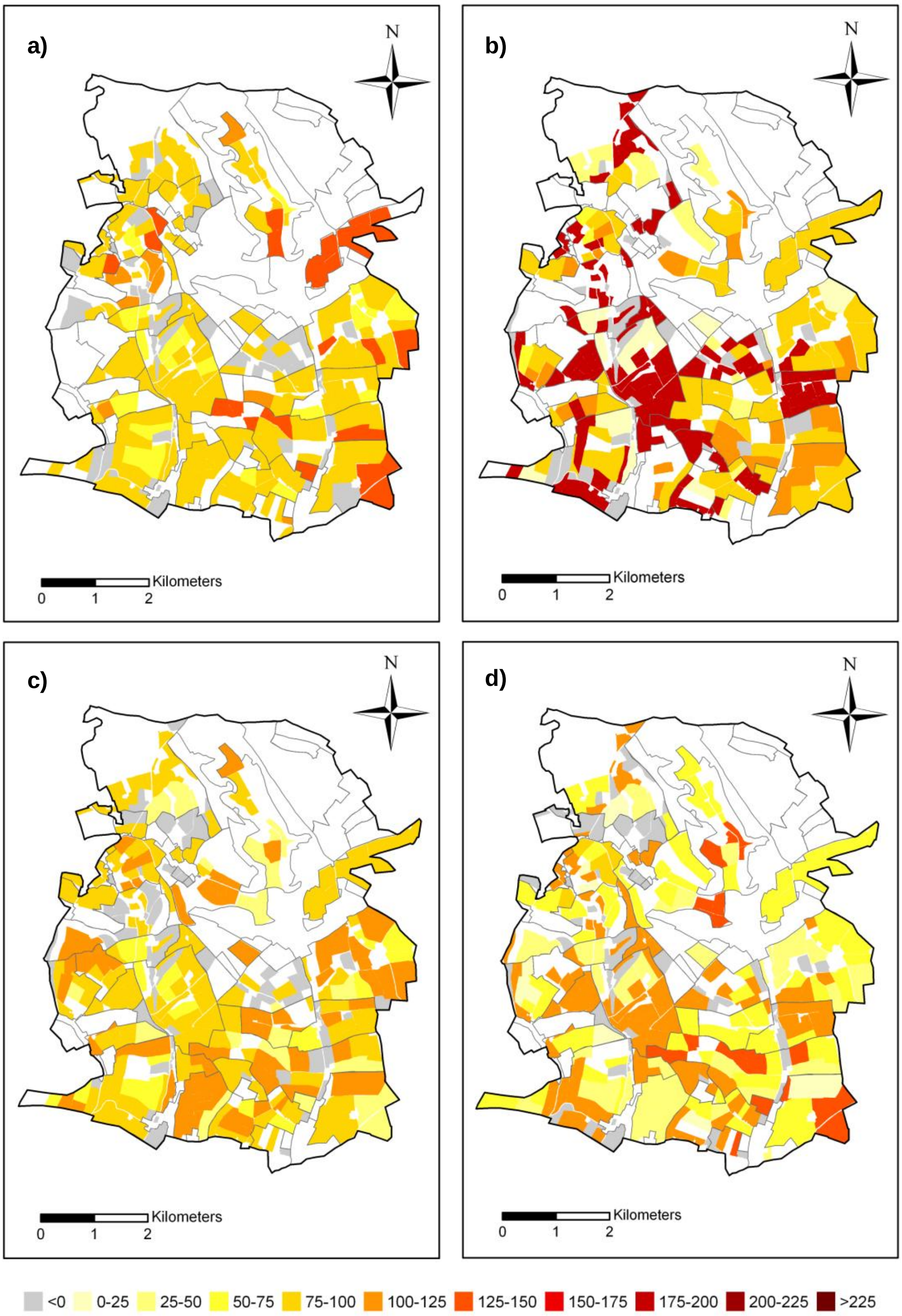


Figure 5-19: Projected field soil surface surpluses (kg N ha^{-1}) for MSA using method 4 in a) 2005, b) 2006, c) 2007, d) 2008.

Limitations of projection methodology

While the projection of field scale results provided an opportunity to investigate possible catchment wide change, methodological limitations have been identified. A lack of field scale results combined with the over (maize and SBM) / under (grass) representation of crops in the field scale subset has compromised the representivity of annual crop average surpluses. Despite grass occupying 40% of MSA and EMEL, grass fields accounted for only 10% of the field scale subset (by area). Grass surpluses were also high, especially in 2006, which when projected over the large grass area made a large contribution to the catchment surplus. Uncertainties were however lessened by the introduction of 0 kg N h^{-1} surpluses to unfertilised grass which occupied 40% of the grass area. Opting for the projection of all year averages may be more appropriate given the sample size; however this provides no indication of mitigation induced change which is key for assessments of mitigation effectiveness.

Projecting surpluses on a crop basis respects the significance of crop type in determining field surpluses. However in doing so field and farm scale variation in nutrient management and environmental conditions are ignored, the effect of which can be observed in Figure 5-20 where actual surpluses are poorly reflected in projected results. In the absence of more data, it may be appropriate to restrict projections to a catchment average which combine farm and field variability captured within the field subset. Field subset selection was however also biased towards fields adopting mitigation, with spring cropped fields and those receiving high nutrient inputs being over represented. Whilst this was appropriate for investigating field scale sensitivity to mitigation, when projected across the whole catchment, there is potential to overestimate mitigation impact.

Despite these limitations, results are not dissimilar from those obtained at the farm and catchment scale (derived using farm scale results – see chapter 6), placing increased confidence in the methodology adopted. Cross scale / method comparisons are discussed in more detail in chapter 8.

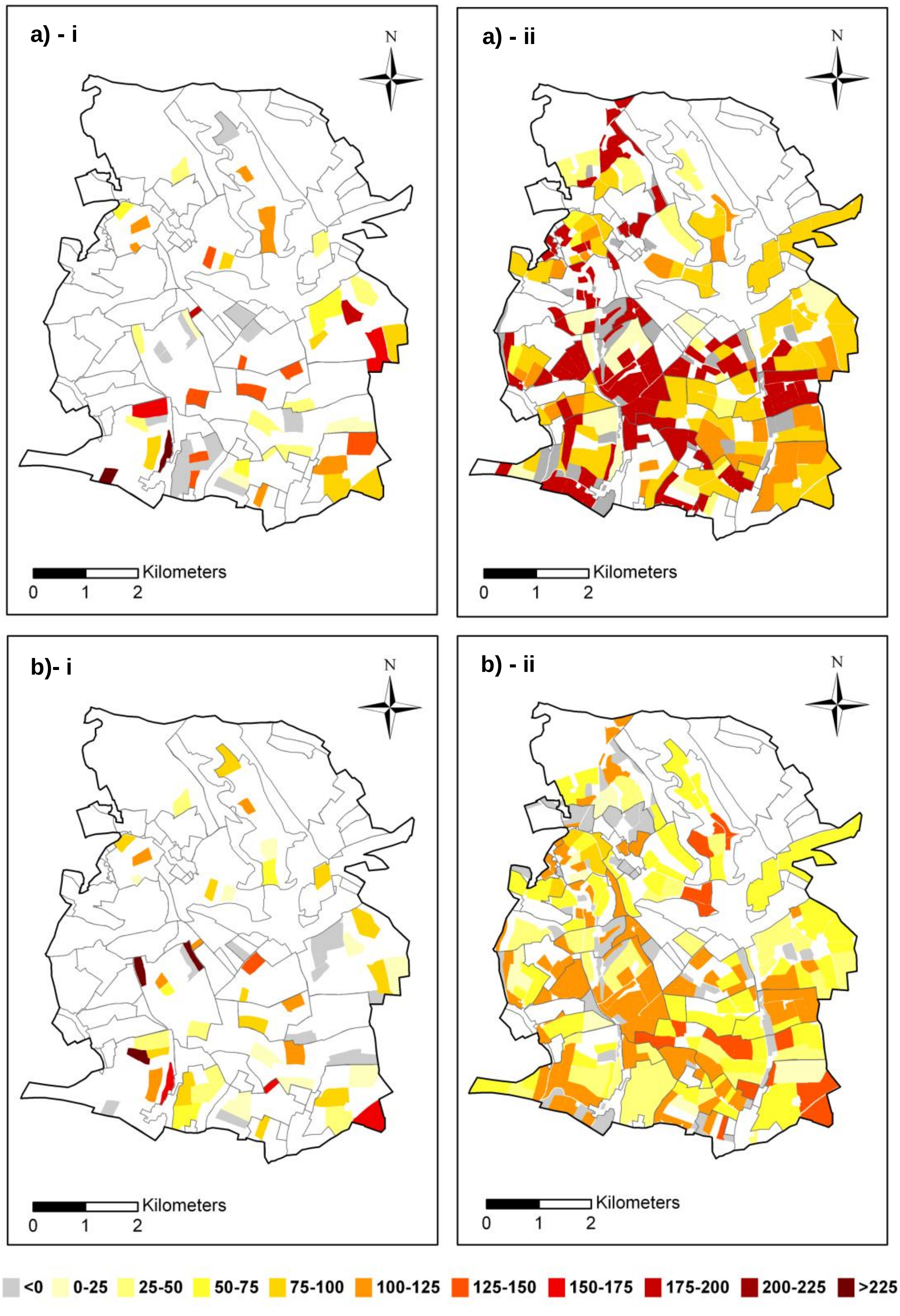


Figure 5-20: Comparison of observed (i) and projected (ii) soil surface balances (kg N ha^{-1}) in a) 2006 and b) 2008.

5.4.3.2 Catchment projection of mitigation scenarios

To assess the catchment scale impact of mitigation scenarios, simulated crop averages where projected across the whole catchment (Figure 5-21). For each scenario adjusted ‘baseline’ crop average surpluses were projected according to 2008 cropping. Similar to the catchment projection of 2005 – 2008 results, area weighted averages were calculated resulting in catchment average surpluses (Table 5-21). It is worth noting that average surpluses were derived from both MSA and EMEL fields and unlike previous catchment projections results are not catchment specific. Mitigation scenarios have been compared to actual 2008 results and to unadjusted ‘baseline’ surpluses projected onto 2008 cropping (to account for differences in cropping between 2005-2007 and 2008).

Table 5-21: Summary catchment projection mitigation scenarios

Mitigation Scenario (see Table 5-10 for details)	Surplus (kg N ha^{-1})		% reductions from 2008 observed		% reductions from 2008 ‘baseline’	
	MSA	*EMEL*	*MSA*	*EMEL*	*MSA*	*EMEL*
1	51.12	48.80	-18.92	-20.88	-36.20	-35.92
2	52.06	49.29	-17.42	-20.08	-35.02	-35.27
3	72.93	69.13	15.68	12.08	-8.98	-9.22
4	82.52	78.38	30.89	27.09	2.99	2.94
5	72.43	68.89	14.88	11.70	-9.60	-9.53
2008 observed	63.05	61.67	0.00	0.00	-21.31	-19.01
2008 ‘baseline’[1]	80.12	76.15	27.09	23.47	0.00	0.00

[1] ‘baseline’ surpluses projected onto 2008 cropping

Significant field scale crop average reductions meant scenarios 1 and 2 had maximum impact on catchment surpluses, with simulated surpluses lower than those observed in 2008. Corresponding with field scale results, scenarios 3, 4 and 5 reduced surpluses relative to ‘baseline’ figures however modelled reductions did not exceed observed reductions post WAgriCo mitigation. Wider applicability of scenario 1 meant improvements were slightly greater than those under scenario 2. Although the latter induced significant surplus reductions in grass and maize, minimal impact on other arable crops and low maize area reduced catchment wide impact. A reduction in manure incorporation delays (scenario 4) increased the catchment surplus above both 2008 and ‘baseline’ results. Scenarios had consistently more effect on EMEL surpluses due to slight differences in crop areas and a lower 2008 surplus from which % change was calculated.

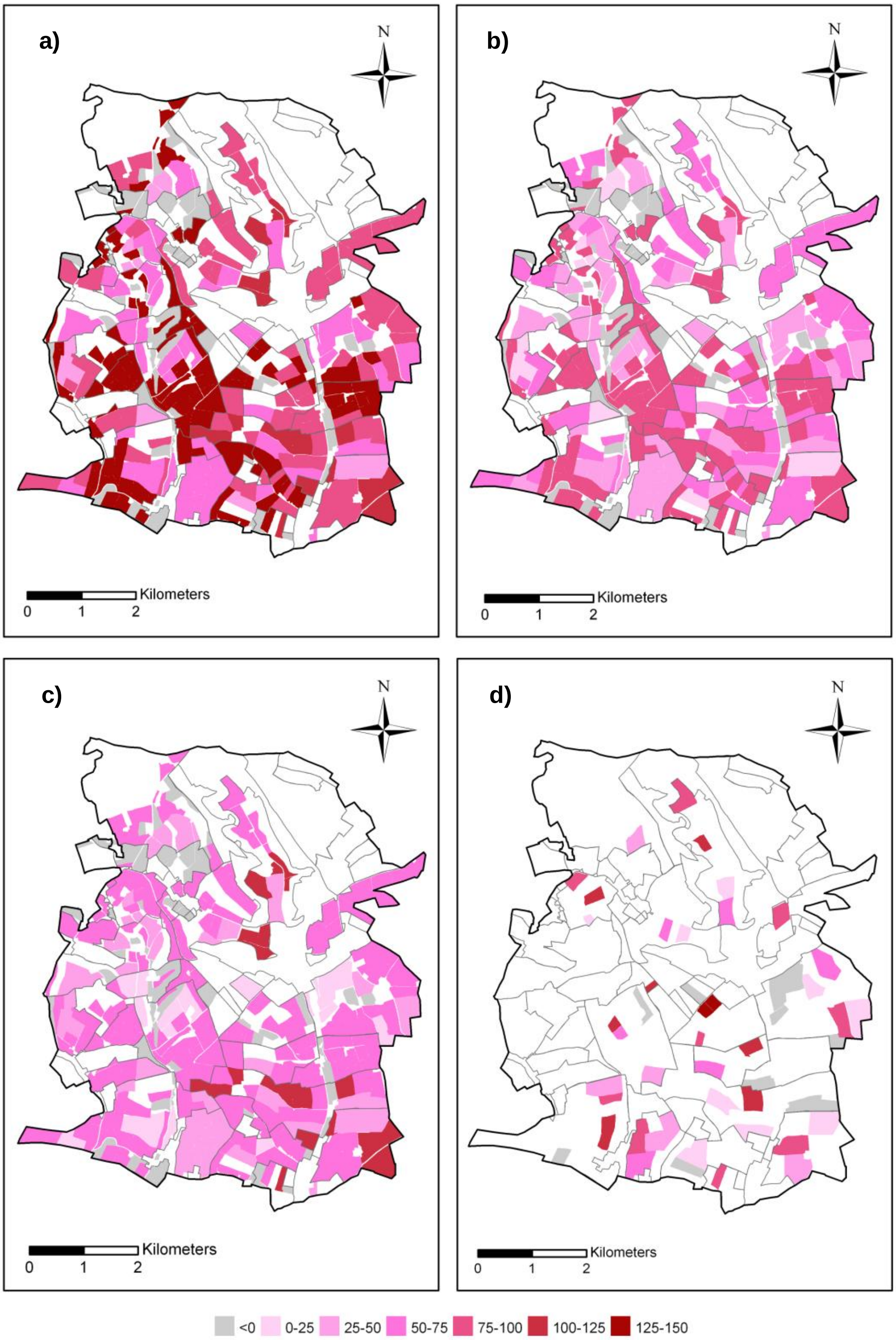


Figure 5-21: Evaluation of mitigation scenarios (kg N ha^{-1}). Comparison of a) baseline projected results (2005-2007 average) with b) mitigation scenario 1 – 20% reduction in fertiliser, c) mitigation scenario 2 – Integration of manure and fertiliser N using TN approach and d) observed mitigation year surpluses.

5.5 Discussion

5.5.1 Sensitivity of soil surface budgets to mitigation in MSA and EMEL

Field scale surpluses tended to decrease following the implementation of mitigation however only in EMEL were differences between pre- and post- mitigation surpluses significant. Improvements in EMEL stemmed from a significant reduction in total inputs, however this did not result from a consistent nor significant reduction in fertiliser or manure across all crops, and only for maize did reliance on a specific nutrient source correspond with a reduction in its use. For example a reduction in fertiliser was not observed for WOSR which was locally reliant on fertiliser inputs. A significant difference in surpluses before and after mitigation was absent from MSA, however a significant reduction in fertiliser was observed for WWF, grass, maize, and SBM. Improvements in fertiliser use were however counteracted by an increased manure inputs, and reductions in total inputs limited to grass, maize and WWM only. Improvements in surpluses were instead driven by significantly higher total output in 2008. Given the timescales over which assessments were made it is important to consider these initial trends relative to other factors affecting inputs and outputs, namely the level of mitigation adopted and external factor for example fertiliser price and weather. Improvements are initially encouraging but may not be linked to mitigation. Indeed changes in MSA surpluses were primarily driven by higher yield which is largely detached from mitigation.

The improvements observed in MSA / EMEL were generally smaller than those reported elsewhere (e.g. Laws et al., 2000 and Korsaeth and Eltun, 2000). Smaller improvements in MSA / EMEL are a likely reflection of modest mitigation induced change in voluntary participation projects like WAgriCo. With farmers risking financial loss, mitigation was unlikely to induce the same degree of change as experimental treatments. Sieling and Kage (2006) for example, investigated the impact of 50% reductions in fertiliser on WOSR, WBF and WW surpluses. Given that fertiliser use in MSA / EMEL decreased by a comparatively modest 5.7% (averaged across all crops) it is not surprising that surplus reductions reported by Sieling and Kage (2006) were more than double those in MSA / EMEL. Assessment of responses to fertiliser recommendations highlighted that fertiliser applications were already lower than recommended levels where only inorganic N was applied, limiting opportunities for voluntary reductions. However where manure was applied, nutrient inputs to maize, SBM, WO, and WOSR were supra-optimal exposing opportunities to reduce fertiliser inputs through improved manure accounting. However only maize received less fertiliser and returned lower total inputs post mitigation, confirming that opportunities for improvement were not fully exploited and there is scope for further reductions in fertiliser in MSA and EMEL. Associated cost savings are likely to be high and should be used to promote better accounting of manure N in fertiliser planning.

The application of manure was found to have a significant effect on field budgets in MSA / EMEL with surpluses up to 164.9kg N ha^{-1} larger where manure was applied. Differences were especially apparent on grass and maize where reliance on manure was at a maximum. The significance of manure inputs on field surpluses was also demonstrated by Sacco et al., (2003) where intensive livestock production on Italian farms meant large manure applications were made to cereals and maize. While in many cases these large surpluses reflect poor accounting of manure N in fertiliser applications, it is important to note the differences in crop availability between fertiliser and manure N. Only some of the total N applied to the soil via manure is readily crop available, meaning larger inputs of organic N are required to satisfy crop nutrient demands than mineral N. However for crops typically receiving large manure applications (maize and grass) reductions in surpluses were larger on fields receiving manure compared to those that did not further supporting the contention that opportunities for improved nutrient management tend to be associated with manure management.

Manure management plans (MMPs) were also found to have a significant and positive effect on field surpluses. Reductions in surpluses were larger where MMP agreements were in place especially for grass, maize and WOSR. A significant impact of MMP on fertiliser, and not manure inputs, suggests responses to MMPs tended to transpire through improved accounting of manure in fertiliser applications. MMPs corresponded with a reduction in manure inputs to maize which is encouraging given the large and at times very large applications of manure made to maize fields in MSA / EMEL. The responsiveness of maize to MMPs supports the large reductions in surpluses on maize fields observed by Sacco et al., (2003) following improvements in manure management on Italian farms. Surpluses decreased by 81% where a maximum manure utilisation scenario was simulated, however the authors note that such large improvements would rely on very efficient exchange of manure between farms.

The crop specific impact of MMPs reflects differences in relative fertiliser / manure reliance. While improvements were not observed in all fields, significant crop x MMP interactions coupled with maximum apparent impact on crops most likely to receive excessive manure applications (grass and maize) suggest MMPs have maximum impact where it is most needed. This is in contrast to fertiliser recommendations which did not appear to address all crops receiving supra-optimal inputs. However some fields on MMP farms continued to receive manure N applications in excess of 400kg N ha^{-1}, demonstrating the flexible nature of MMPs which unlike fertiliser recommendations were interpreted and implemented at the discretion of the farmer. And with uptake of MMPs lower than fertiliser recommendations, excessively large manure input continued to be applied on non MMP farms. In terms of mitigation assessment, evaluation of MMPs at the farm scale has the potential to conceal high risk activity taking place at the field scale. It is important to consider the variability in results to account for high surplus fields where the risk of loss may be disproportionately high.

Surplus reductions tended to be larger where spring crops were preceded by a cover crop, however differences were not significant. Investigations on a farm specific basis confirmed inconsistent responses to cover crops; in some cases cover crops corresponded with a reduction in inputs on some farms, whilst on others fertiliser applications increased. However given that soil surface budgets do not account for leached losses and are insensitive to the immobilisation of available nutrients when the risk of loss is high as demonstrated by Torstensson and Aronsson (2000), the full benefit of cover crops was not expected to be captured by the soil surface balance. Only where cover crops were incorporated and accounted for in subsequent fertiliser recommendations, or utilised through cutting or grazing was a positive response to cover crops expected. While it is difficult to apportion fertiliser reductions to cover crops and fertiliser recommendations respectively, the insignificance of this result suggests there remains opportunities to account for crop residues and to utilise cover crops more effectively. The positive response observed across all fields may reflect more substantial responses to other mitigation methods implemented on more environmentally aware farms i.e. those willing to plant cover crops.

Uptake of ‘spring manure applications’ were low with only 1 farm agreeing to the measure. While result should therefore be treated with caution, they were, however, encouraging. A lower risk of manure N leaching in the spring (Beckwith et al., 1998; Chambers et al., 2000; Smith et al., 2002) meant delayed manure applications were smaller than those made in previous autumns, and not compensated by increased fertiliser inputs. Surpluses fell by 67.4kg N ha^{-1} between pre- and post- mitigation years highlighting the sensitivity of budgets to timing based mitigation despite being calculated on an annual basis. (It was previously assumed that budgets calculated on annual basis would not capture a change in the date of management activities. However these results have demonstrated that where a change in the timing results in a change in the amount of an input / output, budgets may be more sensitive to 'timing' based mitigation than previous thought). Irrespective of these positive findings, low uptake questions the applicability and favourability of this measure.

Spatial representation exposed surplus hotspots and accounted for field / farm specific management, however on an annual basis it is unclear whether hotspots reflect cropping or poor nutrient management. Averaging surpluses across complete crop rotations would better account for cropping (this was not conducted because crop rotations extended beyond the four year project). Longer term surpluses, similar to those calculated in Berry et al., (2003) better reflect strategic crop interactions and the longer term release of organic N to the available phase. Calculated on an annual basis, manure inputs contribute only to the balance of the current crop despite being of benefiting to subsequent crops.

Improvements in surpluses reflect changes in inputs and outputs that may or may not have stemmed from the implementation of mitigation. In the absence of a control treatment it is important that results are interpreted with respect to economic and environmental change. Fertiliser use decreased nationally between pre- and post-mitigation years, with reductions ranging from 1.2% on WOSR to 9.2% on maize (BSFP, 2009). Although maximum and minimum surplus reductions in MSA / EMEL corresponded with maximum / minimum reductions in fertiliser use nationally, patterns of annual variability showed little resemblance (gradual decrease vs. higher applications in 2005 and 2007 – see Figure 5-4). Moreover where MSA / EMEL surpluses fell, corresponding reductions in fertiliser exceeded those observed nationally suggesting fertiliser price was not the main driver of change. Having said this, fertiliser reductions in MSA and EMEL were not restricted to 2007/8 and in some cases applications increased between these years despite a reduction in use between pre- and post- mitigation year usage.

Yield was not significantly affected by year, however initial downward trends in surpluses on MSA grass, maize, SBM, WBF and WWF fields were primarily driven by a significant increase in yield post mitigation. Results therefore expose an appreciable link between environmental condition and field balances. Longer studies are required to provide greater assurance that change is in response to mitigation programmes. In the Netherlands and Denmark for example, surpluses were evaluated over 13 / 14 years to investigate the impact of Mineral Policy and National Action Plans (Kronvang et al., 2008; Zwart et al., 2008). However where time scales are short, evaluating changes in inputs independent of output, which the farmer has less control over, may prove more informative and objective. In terms of nutrient inputs, speaking directly to farmers is the only way to understanding reasons for changes in nutrient inputs. Discussions with farmers would also provide an opportunity to differentiate between responses to different mitigation options. Responses to a range of mitigation methods were captured by a limited number of inputs / output (namely fertiliser inputs) making it difficult to objectively assessment the impact of individual mitigation methods.

5.5.2 Field scale mitigation scenarios

Field scale mitigation scenarios highlighted the effectiveness of reducing fertiliser inputs across all crops, resulting in consistently high surplus reductions of 30 – 60%. However given that 'baseline' fertiliser inputs in MSA / EMEL were less than recommended levels, and observed reductions in fertiliser following WAgriCo mitigation generally smaller than simulated reductions, such large reductions are unlikely to occur voluntarily. For crops reliant on manure, namely grass and maize, maximum accounting of manure N was more effective than fertiliser reductions. However, given that only ammonium / uric acid are readily available for crop uptake, accounting for manure N on a total N basis is unlikely to fulfil crop N requirements in the short term. Accounting for N on a crop available basis provides greater certainty of crop N supply although improvements were considerably smaller. With improvements attributed to MMPs occupying an intermediate position between those of the two manure accounting scenarios, observed results confirm that improved manure accounting has the potential to significantly reduce surpluses even on a voluntary basis. In contrast to fertiliser based mitigation, manure scenarios redress the balance of inputs and outputs rather than simply reducing inputs. As such yields are not jeopardised and unnecessary fertiliser costs avoided.

While observed responses to WAgriCo mitigation and modelled results are of a similar magnitude increasing confidence in the conclusions drawn from the latter, scenarios afford no consideration to compensatory behaviour (i.e. an increase in one input in response to a reduction of another) or negative impact on output. Reductions in fertiliser inputs may in reality be compensated by increased manure applications or result in reduced yield, and where nutrient inputs to grass effects grazing potential, reduced grass production may be compensated by increased feed import. The absence of feed in field scale balances may support the use of farm scale evaluations to capture change in all nutrient flows.

5.5.3 Sensitivity analysis

Sensitivity of the field balance to inputs and coefficients highlighted where uncertainty in farm data and unrepresentative coefficients is most likely to transpire in surpluses. Accurate input data and appropriate coefficients are especially important where the balance displays high sensitivity. In agreement with Campling et al., (2005) field scale budgets displayed maximum sensitivity to fertiliser inputs and crop related factors (yield and crop N) with % changes in surpluses exceeding adjustments in inputs / coefficients. High sensitivity stemmed from substantial fertiliser inputs to all crop and crop offtake, the product of yield and crop N, representing the only balance output. Sensitivity to manure inputs / coefficients reflected reliance on manure, with maximum sensitivity observed on maize due to high organic N inputs.

Although sensitivity to fertiliser inputs was high, statutory requirements that fertiliser applications be recorded in NVZs meant input data was reliable and complete. In contrast, variability in measured crop N content and concerns surrounding the accuracy of 2008 yields means sensitivity, uncertainty and variability coincide in balance output. Crop N sampling was however conducted on an ad-hoc not representative basis, and despite high variability ‘baseline’ surpluses were within the range of those calculated using local crop N contents with the exception of WOSR. In terms of yield, estimates provided in 2008 may overestimate the quantity of N removed from the soil, exaggerating the impact of mitigation. However yields were not significantly different between years, and crop coefficients induced a systematic error, meaning impact on mitigation effectiveness evaluations was minimal. Sensitivity of high offtake crops supports the prioritisation of precise crop N values over manure N values as suggested by Campling et al., (2005). Although manure data was often incomplete and measured manure N contents variable, only maize was sensitive to manure inputs / coefficients. Moreover, observed improvements on maize exceeded those following a simulated 20% reduction in manure inputs, increasing confidence in the mitigation evaluations performed. However budget calculations involved all aforementioned factors, and on a cumulative basis uncertainty may still exceed responsiveness to mitigation. Indeed improvements in surpluses following WAgriCo mitigation were smaller than the changes induced by a 20% adjustment of at least 2 coefficients / variables emphasising the need for accurate and complete input data and coefficients.

5.5.4 Catchment scale soil surface surpluses

Catchment scale surpluses were consistently lower following the implementation of mitigation with reductions ranging from 19.6 to 28.1%. Smallest reduction were observed in MSA where calculations were made using MSA specific farm data only and results not enhanced by the larger reductions observed in EMEL. Improvements were therefore of a similar magnitude to those seen at the field scale, however over representation of fields subject to field scale mitigation may have exaggerated catchment scale improvements. Despite variable year on year ranks between catchments and calculation methods, 2008 results consistently represented the lowest surplus. However catchment scale surpluses reflect both cropping and nutrient management meaning improvements cannot be directly linked to mitigation.

With catchment surpluses calculated on a crop type basis, results were extrapolated to individual fields based on annual cropping. Although an improvement in 2008 was evident, moderately high surpluses remained in some fields in 2008. Catchment average surpluses have the potential to conceal surplus hotspots, and fail to inform whether such fields have been effectively addressed by mitigation. Comparison of actual and projected field scale results highlights limitations of the extrapolation process used to provide these more informative field scale evaluations. Discrepancies between projected and actual results expose the magnitude of field and farm variability which is not captured by ‘crop type’ projections. Furthermore, projected results assume uniform mitigation induced change which does not reflect the farm and field specific nature of mitigation implementation. Although catchment surpluses offer a less detailed means of evaluation, their coarse resolution means all variation at farm and field scale is effectively captured. Reliability depends only on the representivity of field subsets. However, where hotspots are likely to exist and variability in field / farm nutrient management / mitigation implementation high, evaluations would benefit from the calculation of surpluses at field or farm scale, but with this comes increased time and cost.

5.5.5 Catchment scale mitigation scenarios

Application of fertiliser but not manure to all crops meant the reduced fertiliser scenario had greater impact at the catchment scale than the TN manure accounting scenario. This was despite larger impact of manure accounting on field scale surpluses (for those crops receiving manure). Low catchment coverage of maize, for which largest reductions were observed, reduced impact at the catchment scale highlighting the importance of mitigation applicability on catchment impact. However by evaluating mitigation at the catchment scale localised but large responses to mitigation on high risk fields are concealed. While maize covers only a small portion of MSA and EMEL (c.5%) it is to these fields that very large manure applications have been observed and thus where a substantial response to mitigation is required.

In agreement with field scale results, simulated reductions in catchment surpluses were larger than those observed in MSA / EMEL (36% vs. 21 and 19% in MSA and EMEL respectively); simulated reductions in fertiliser were larger than those observed in MSA / EMEL and manure N accounting not yet maximised. Responses to fertiliser scenarios were however in line with those reported by Gomann et al., (2005) where a hypothetical fertiliser tax and subsequent reduction in fertiliser use reduced catchment surpluses by 27 and 34% in the Ems and Rhine catchments. Gomann et al., (2005) also commented on the wider applicability of fertiliser based mitigation, but, in accordance with MSA/EMEL results, acknowledged its limitations where manure production / use was high (e.g. maize). Due to high ‘baseline’ surpluses and large reductions in fertiliser, Eulenstein et al., (2008) observed slightly larger reductions of 40 to 56%. However as previously noted, such large reductions in fertiliser were unlikely to occur voluntarily in MSA / EMEL. Improvements in MSA / EMEL were less than those under the maximum manure accounting scenario (but more than under the conservative RAN approach). Farm specific investigations exposed opportunities for continued improvement in manure accounting especially on non MMP farms, however realisation of a 35% surplus reduction depends on the extent to which manure is already accounted for under 'baseline' practices and the ability of manure total N to supply readily available N. The scenario assumes no prior accounting of manure N meaning impact is overestimated on farms where manure nutrient management is already good.

Where results have been projected back to the field scale, they suffer from the same uncertainty attached to non mitigation projections in addition to the uncertainties attached to scenario assumptions. While catchment surpluses do not reflect where mitigation had most impact, they avoid the complications of field and farm variability. As such projections should be restricted to providing a visual interpretation of results for comparative purposes, not offering field specific responses to mitigation. Given the simplified, no feedback system used to simulate mitigation, and the uniform 'baseline' conditions assumed, scenarios provide an indication of maximum change. Whether such large improvements can be observed depends on the degree of change, compensatory behaviour and the availability of funding / regulation to compensate / enforce implementation.

5.6 Conclusions

Field scale budgets displayed a tendency towards lower surpluses and fertiliser use post mitigation suggesting field budgets captured initial improvements in nutrient management particularly in EMEL. However inconsistencies and insignificant responses suggests WAgriCo mitigation had modest impact in nutrient management especially in MSA where improvements were largely driven by higher yield post mitigation. Sensitivity to differences in nutrient use between crops and catchments, and in some cases years, does however suggests that if mitigation were to induce greater change, field budgets would capture improvements and thus provide an initial indication of mitigation impact.

Results demonstrated the difficulties, but also highlighted the realities, associated with catchment scale projects compared to designed experiments in which the degree of change reflects 'baseline' management, incentives and compensation, and level of farmer discretion through which mitigation is implemented. Indeed farm data highlighted opportunities for further improvements despite the implementation of mitigation targeting the poor management in practice with large manure applications on some MMP farms and supra-optimal nutrient inputs despite the adoption of fertiliser recommendations. Fluctuations in fertiliser price and sensitivity to weather via yield added further uncertainty and variability to results questioning whether improvements were connected to mitigation. More comprehensive assessments of budget sensitivity to mitigation require longer term investigations and more drastic mitigation to account for some of the uncertainties associated with annual variability. However speaking directly to farmers is the only way to fully understand why changes have been made, and where yield fluctuates significantly, it would be beneficial to look at changes in inputs independent of output.

Field scale budgets failed to capture the benefits associated with cover crops however this was not surprising given that budgets poorly account for the immobilisation of N during times of high leaching. Field budgets did however confirm the effectiveness of manure management plans at reducing the risk of N loss from fields receiving manure through improved manure accounting. Mitigation scenarios and analysis of farm data relative to fertiliser recommendations further supported these findings exposing opportunities for improvements in manure management beyond those observed under WAgriCo mitigation. Field scale budgets showed that while WAgriCo mitigation may not have fully addressed the nutrient management issues in EMEL and MSA in the short term, the mitigation and assessment methods implemented were effective in exposing the problems and highlighting opportunities for improvement.

The impact of MMPs was found to be crop specific demonstrating maximum impact where it is most needed (on fields most likely to receive excessively large manure applications for example maize). However comparison of field and catchment scale mitigation scenarios highlighted the need to consider uptake and applicability across the catchment; with improved manure management likely to affect a smaller proportion of fields than reductions in fertiliser. But conversely, adherence to fertiliser recommendations on most farms meant further reductions would require financial compensation whilst improvements in manure management bring direct financial returns. Comparison of field and catchment scale methods also highlighted the absence of hotspots in the latter, and the insensitivity of catchment scale methods to capture localised but disproportionately significant improvements where mitigation addresses these hotspots. Evaluations would however benefit from longer term assessment to ensure strategic interactions between crops and opportunities for mitigation across crop rotations are fully accounted for. Once considered the identification of hotspots is useful for the directing and targeting of future mitigation.

6 Using farm scale nutrient budgets to evaluate mitigation effectiveness

6.1 Introduction

6.1.1 Introduction to farm scale nutrient budgets and mitigation evaluation

6.1.1.1 Nutrient budgets and their relevance to nutrient loss and mitigation evaluation

Nutrient budgets are commonly used to evaluate inputs and outputs of nutrients in agricultural systems, exposing nutrient surpluses / deficits and highlighting areas of inefficient / efficient nutrient utilisation. Where inputs are not effectively converted into useful output such as crops, milk or meat, the resulting nutrient surpluses are at risk of loss to the environment. Although the relationship between surpluses and nutrient loss is affected by climate and soil conditions, and is often considered indirect, a reduction in nutrient surpluses is likely to yield environmental benefits. Mitigation methods aimed at reducing nitrate loss have been developed to minimise the environmental impact of agricultural activities. Many aim to redress nutrient imbalances thereby exposing an inherent link between nutrient budgets and mitigation. Comparing farm budgets before and after the implementation of mitigation provides an opportunity to assess the effectiveness of these actions in reducing nitrate (N) loss to water.

6.1.1.2 Nutrient budgets at the farm scale

Nutrient budgets can be calculated at a range of scales but are most commonly applied to the farm scale. Farm scale budgets enable the farm system to be modelled as a whole, respecting the complex interactions and feedbacks which occur especially on mixed farms (Halberg et al., 2005). Inclusion of all areas of the farm means that significant nutrient transfers occurring within livestock housing via feed, are fully accounted for. Where budgets are calculated at the field scale such nutrient sources are rarely considered even where surpluses from across the farm are summed together. For the benefit of mitigation evaluations, farm scale budgets effectively account for compensatory behaviour; for example a reduction in fertiliser to grass may result in increased feed imports. The whole system approach means the benefits and opportunities for improved internal nutrient cycling are effectively exposed (Brouwer, 1998; Oenema et al., 2003). By avoiding the inherently different nutrient use efficiency attached to specific crops (which affects results at the field scale – see chapter 5), farm scale budgets provide a more objective means of assessing nutrient management and the impact of mitigation (Lord et al., 2002). Investigations need not span complete crop rotations to expose the influences of nutrient management over cropping, facilitating shorter term investigations. Farm scale investigations respect the differences in nutrient use between farm types (Domburg et al., 2000; Ondersteijn et al., 2002; Nielsen and Kristensen, 2005; Bassanino et al., 2007), acknowledging the inefficient utilisation of nutrients in livestock systems.

Farm scale budgets are calculated at a scale relevant to farmers, providing assurance that changes imposed by new policy are compatible with existing farm systems (Oenema et al., 2003; Schroder et al., 2003; Goodlass et al., 2003). Farm budgets have the potential to raise awareness of nutrient use, highlight inefficient nutrient transfers, and motivate and facilitate better environmental performance (Goodlass et al., 2003, Halberg et al., 2005). They also reflect the level at which most operational and management decisions are made, and the level at which mitigation is likely to be decided and applied (Van Beek et al., 2003). It is therefore appropriate to investigate changes in nutrient management and the impact of mitigation at this same scale.

Data requirements and levels of uncertainty further support the use of farm scale methodologies (Oenema et al., 2003). Farm budgets are typically easier to calculate and unlike field scale budgets rarely demand information on the management of individual crops (Bassanino et al., 2007). Data requirements are therefore relatively easy to satisfy and can be supplied with a higher level of certainty. The calculation of farm scale budgets over field scale methodologies is therefore encouraged (e.g. Lord et al., 2002), contributing to the widespread use of farm scale budgets in agri-environmental contexts. While this has failed to result in consistent and standardised methodologies and reference values (Oborn et al., 2003; Oenema et al., 2003), the large body of literature means that results can be placed in a wider context, and greater confidence held in the adoption of specific methodologies.

6.1.1.3 Farm scale budgets and mitigation

Farm scale budgets are considered an indicator of agri-environmental performance and sustainability (Oenema et al., 2003; Schroder et al., 2003; Bassanino et al., 2007) and are commonly used to evaluate the effect of agri-environmental legislation and monitor changes in nutrient use. For example Zwart et al., (2008), Kyllingsbaek and Hansen, (2007) and Verbruggen et al., (2005) report temporal variability in national nutrient surpluses in The Netherlands, Denmark and Belgium respectively, relating change to the introduction of national policy aimed at reducing the environmental impact of agriculture. The availability of data requirements enables large scale applications of farm scale budget methodologies. Farm scale budgets have themselves also been implemented as a policy measure, acknowledging their ability to engage farmers (Goodlass et al., 2003; Halberg et al., 2005) and the environmental benefits of surplus reductions (Oenema et al., 2005; Kyllingsbaek and Hansen, 2007). In both the Netherlands and Switzerland, target nutrient surplus / reductions have been utilised as a means of reducing nutrient loss (Ondersteijn et al., 2002; Hanegraaf and den Boer, 2003; Herzog et al., 2008). In New Zealand the farm budget based model OVERSEER (Wheeler et al., 2003) fulfils a regulatory role, calculating ‘nitrogen discharge allowances’ below which future farming must operate (Shepherd et al., 2009); exact timescales and approaches differ between regions (Shepherd, pers. comm.). While there have been some concerns surrounding legislative compliance, budget based mitigation is responsive and attributable, and allows farmers to decide themselves how best to meet targets within their specific farm system (Ondersteijn et al., 2002; Hanegraaf and den Boer, 2003; Schroder et al., 2003).

Given the adoption of budget based policy and the likely benefits of lower surpluses, ways of reducing surpluses have also been explored. Nutrient budgets have been applied to demonstration farms to assess the impact and practicalities of mitigation options on farm nutrient surpluses (Aarts et al., 2000). Subsequent studies have investigated how commercial farms can achieve similar surplus reductions (Oenema et al., 2001; Langeveld et al., 2005). Although not the main purpose of these applications, such studies indirectly offer an evaluation of mitigation effectiveness. However, the goal driven nature of this mitigation approach means target surpluses remain the focal point and less attention is afforded to the specific means by which this is achieved. Investigations are unlikely to focus on the impact of specific mitigation methods and how this is affected by farm type, as is required for mitigation evaluations. Examples of explicit assessments of mitigation impact on farm budgets are much rarer, exposing an opportunity to explore this avenue further. Kuipers and Manderloot (1999) modelled the likely surplus reductions induced by a range of individual mitigation methods and although the work is encouraging, a larger, more recent evidence base is required to objectively assess the applicability of budgets to mitigation evaluation. Investigations based on observed farm data are needed to confirm the sensitivity of budgets to current mitigation methods under more recent economic and regulatory circumstances.

Continued use at the farm scale and inherent sensitivity to changes in inputs / outputs means farm scale budgets have the potential to simply and effectively evaluate mitigation effectiveness. It is on these grounds that further investigation into the use of farm scale nutrient budgets as an evaluator of mitigation effectiveness is proposed. Given the 'waterbody' focus of the WFD it is also important to consider the applicability of farm scale budgets to the wider catchment. Extending farm scale methodologies across the whole catchment is vital to ensure assessment methods are relevant under current legislation. Investigations exploring the sensitivity of farm scale budget methodologies applied at farm and catchment level are required to assess their suitability for mitigation evaluation and the extent to which they address legislative demands.

6.1.2 The current study

6.1.2.1 Hypothesis 1

'Farm scale nutrient budgets represent an effective method of mitigation evaluation.'

6.1.2.2 Aim

To assess the sensitivity of farm scale nutrient budgets to a range of mitigation methods at the farm and catchment scale.

6.1.2.3 Objectives

1. To investigate farm scale nutrient budget methodologies and subsequently develop or adopt an appropriate method for the purpose of mitigation evaluation.
2. To explore farm scale nutrient management and investigate associated spatial variability of nutrient utilisation.
3. To investigate the impact of mitigation on nutrient surpluses by:
    a. Comparing farm nutrient surpluses before and after the implementation of mitigation on MSA and EMEL farms.
    b. Comparing responses to different levels of mitigation
    c. Simulating mitigation scenarios to extend the range and magnitude of mitigation employed.
4. To investigate the applicability of farm scale methodologies for the assessment of catchment scale mitigation impact by:
    a. Upscaling the farm scale methodology to the catchment scale
    b. Comparing catchment surpluses before and after the implementation of mitigation in MSA and EMEL.

6.2 Method

6.2.1 Farm scale budget methodologies

Despite widespread acceptance and applicability of farm nutrient budgets, there remains no internationally accepted standard methodology (Goodlass et al., 2003). Most adopt a farm gate approach, quantifying nutrients that enter and leave the farm gate, offering no consideration to internal transfers or loss processes (Figure 6-1). Inputs typically include fertiliser, feed, animal imports, and manure imports whilst animal, manure, crop and animal product exports are included as outputs (see Table A-2 in the appendix). Popularity is fuelled by their relatively low and easily obtainable data requirements, and a lower level of uncertainty (Watson and Atkinson, 1999; Lord et al., 2002; Oenema et al., 2003). By treating the farm system as a 'black box' the use of manure is not explicitly accounted for, however difficulties in obtaining representative manure N contents and assessing grass yield / grazing offtake are avoided (Lord et al., 2002; Oenema et al., 2003). Whole system balances provide the main alternative to farm gate methodologies, applying a soil system balance to the entire farm system, however doubts surround the appropriateness of such approaches to highly complex farm systems (Watson and Atkinson, 1999). As an intermediate option, some budgets apportion surpluses to different loss pathways (Ledgard et al., 1999) while others calculate balances for individual system components which are pieced together to create a picture of the whole farm's nutrient use (Jurgen et al., 2006).

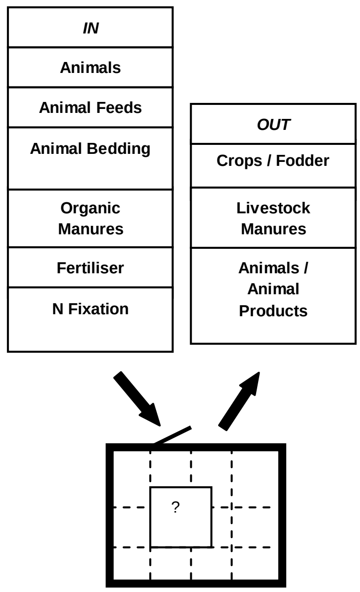


Figure 6-1: The farmgate budget methodology (figure adapted from Oenema et al., 2003)

6.2.2 Developing a farm scale methodology

In keeping with previous farm scale nutrient budget studies, the availability of input data meant a farmgate approach was favoured, and with numerous previous applications of farm scale budgets, a suitable methodology for mitigation evaluation was likely to already exist. UK approaches were investigated and the PLANET methodology (Defra, 2005) highlighted for further investigation owing to its recent and comprehensive development. The PLANET budget was developed for Defra following an appraisal of existing nutrient budget methodologies and aimed to provide a straightforward, standard nutrient audit methodology to assess nutrient utilisation performance. Data requirements were typical of farmgate budgets and corresponded with the availability of WAgriCo farm data. Despite not passing through the farm gate N fixation was included on the grounds that planting legumes represent a strategic nutrient management decision. In contrast atmospheric deposition was excluded on the grounds that it is outside the farmer's direct control. Smaller inputs such as seed, irrigation and stock water excluded for simplicity. The methodology was previously used in a national study to investigate nutrient flows within different farm types and to develop benchmarks to assess farm performance (Defra, 2005). The study confirmed the methodologies applicability of to UK farming systems and ability to characterise nutrient use.

Preliminary calculations were conducted to ensure the methodology was applicable to local farming systems, and that the exclusion of 'smaller' inputs and atmospheric deposition, and inclusion of N fixation was justified within the study catchments. Preliminary budget calculations for cereal and dairy farms confirmed that seeds contributed very little N to the balance and did not justify the additional input data and computation. Atmospheric deposition represented a significant input but derived from a fixed, area weighted input, had little impact on before and after comparisons. In contrast, N fixation represented a large input on legume dependant farms; for example, where dairy farms have large areas of clover rich grass. Although not strictly entering via the farm gate, the significance of this input and the conscious decision to source N in this way supports its inclusion. These findings were in agreement with the PLANET project, and supported the direct application of the PLANET methodology to the current study.

6.2.3 Calculating the farm scale budget

Farm gate budgets were calculated annually, by harvest year, using farm data supplied by the WAgriCo project (Table 6-1). Budgets were performed using an Excel spreadsheet and results presented on a 'per ha' basis; details of the calculations undertaken are provided in Table 6-2. Worked examples for cereal and dairy farms are presented in Table 6-3 and Table 6-4. Budgets were calculated for all MSA and EMEL farms for which input data was available and mitigation adopted in the final year of the study. This represented a maximum of 34 farms in any one year (see Table 3-3).

Table 6-1: Farm data collected annually during the WAgriCo Project (2005 – 2008)

Input / output	Data field	Availability
Inputs	Imported fertiliser (kg N) [a]	High
	Imported livestock (no. and weight (kg))	Medium / Low[b]
	Imported animal feed (t)	High
	Imported fodder (t)	Medium
	Imported organic manures (t)	Low - high
	Imported animal bedding (t)	Medium
	Area of legumes (ha)	Low - medium
Outputs	Exported livestock (no. and weight (kg))	Medium / Lowb
	Exported crops (t)	High
	Exported fodder (t)	Medium
	Exported straw (t)	Medium
	Exported organic manure (t)	Low - high
	Exported animal products (milk / wool / eggs) (l / kg)	High
n/a	Farm area (ha)	High

[a] Σ fertiliser inputs to each field

[b] Low for livestock weight

Table 6-2: Calculating the PLANET farmgate budget

INPUTS	Calculation details	References
Livestock	Σ(No. animals x weight (kg) x livestock N content (kg N t^{-1} fw) / 1000)	Various – see PLANET report (Defra, 2005)
Feed	Σ(Concentrate / fodder (t) x feed N content (kg N t^{-1} fw))	
Animal bedding	Σ(Bedding (t) x bedding N content (kg N t^{-1} fw))	
Manure	Σ(Manure (t) x manure N content (kg N t^{-1} fw))	
Fertiliser	Σ(Fertiliser (t) x fertiliser N content (%) / 100)	
N fixation	Σ(Area of legume (ha) x fixation rate (kg ha^{-1}))	
Total Inputs	Σ Inputs	

OUTPUTS	Calculation details	References
Livestock	Σ(No. animals x weight (kg) x livestock N content (kg N t^{-1} fw) / 1000)	Various – see PLANET report (Defra, 2005)
Animal products	(No. eggs x weight (kg) x egg N content kg N t^{-1} /1000) + (milk (l) x N content (kg N t^{-1}) /1000) + Σ(wool (kg) x N content (kg N t^{-1}) /1000)	
Crops	Σ(Total crop export (t) x N content (kg N t^{-1} fw))	
Manure	Σ(Manure (t) x manure N content (kg N t^{-1}fw))	
Total Outputs	Σ Outputs	

BALANCE	Calculation details	References
	(Total inputs – Total outputs) / farm area (ha)	n/a

Table 6-3: Worked example of a 'PLANET' farmgate budget for a cereal farm

INPUTS	Farm data	Calculation		Kg N
Livestock	None	n/a	n/a	0
Feed	10t Beef calf feed, 300t grass silage, 30t feed wheat	Σ(Concentrate / fodder (t) x feed N content (kg N t^{-1} fw))	(10 x 28.8) + (300 x 6.8) + (30 x 17.9)	2863.5
Animal bedding	None	n/a	n/a	0
Manure	None	n/a	n/a	0
Fertiliser	45141.2kg fertiliser N	Σ(Fertiliser (t) x Fertiliser N content (%) / 100)	45141.2 x 100 / 100	45141.2
N fixation	None	n/a	n/a	0
Total Inputs		Σ Inputs	2863.5 + 45141.2	48004.7
OUTPUTS	**Farm data**	**Calculation**		**Kg N**
Livestock	30 grower / fatteners 12-24months	Σ(No. animals x weight (kg) x livestock N content (kg N t^{-1} fw) / 1000)	30 x 400 x 22.5 / 1000	270.0
Animal products	None	n/a	n/a	0
Crops	563t feed wheat, 614t malting barley, 366t oats, 190t oilseed rape	Σ(Total crop yield (t) x N content (kg N t^{-1} fw))	(563 x 17) + (614 x 14) + (366 x 17) + (190 x 30)	30089.0
Manure	None	n/a	n/a	0
Total Outputs		Σ Outputs	270.0 + 30089.0	30359.0
BALANCE	(Total inputs – Total outputs) / farm area (ha)			**70.7 kg N ha^{-1}**

Table 6-4: Worked example of a farmgate budget calculation for a dairy farm

INPUTS	Farm data	Calculation		Kg N
Livestock	16 dairy cows	Σ(No. animals x weight (kg) x livestock N content (kg N t^{-1} fw) / 1000)	16 x 500 x 22.5 / 1000	180.0
Feed	90t 25% CP dairy feed, 90t 18% dairy feed, 350t grass silage, 300t maize silage	Σ(Concentrate / Fodder (t) x feed N content (kg N t^{-1} fw))	(90 x 40) + (90 x 28.8) + (350 x 6.8) + (300 x 4.8)	10010.2
Animal bedding	80t wheat straw	Σ(Bedding (t) x bedding N content (kg N t^{-1} fw))	80 x 5	400.0
Manure	None	n/a	n/a	0
Fertiliser	4714.2t fertiliser N	Σ(Fertiliser (t) x Fertiliser N content (%) / 100)	4714.2 x 100 / 100	4714.2
N fixation	None	n/a	n/a	0
Total Inputs		Σ Inputs	180 + 10010.2 + 400 + 4714.2	15304.4
OUTPUTS	**Farm data**	**Calculation**		**Kg N**
Livestock	30 dairy cows, 70 calves sold at 12 weeks	Σ(No. animals x weight (kg) x livestock N content (kg N t^{-1} fw) / 1000)	(30 x 500 x 22.5 / 1000) + (70 x 80 x 22.5 / 1000)	463.5
Animal products	580,000l milk	Σ(milk (l) x N content (kg N t^{-1}) /1000)	580,000 x 5 / 1000	2900.0
Crops	None	n/a	n/a	0
Manure	158t cattle FYM	Σ(Manure (t) x Manure N content (kg N t^{-1}fw))	158 x 6	948.0
Total Outputs		Σ Outputs	463.5 + 2900 +948	4311.5
BALANCE	(Total inputs – Total outputs) / farm area (ha)			**353.5 kg N ha^{-1}**

6.2.4 Data interpretation

To investigate the sensitivity of farm gate budgets to mitigation, results (inputs, outputs and surpluses) were compared before and after the implementation of mitigation, and responses to different levels of mitigation (GAP and EGAP) explored; 2005 – 2007 results were averaged to produce a 'before' dataset, whilst post-mitigation results were based on 2008 results only. To confirm the methodology's robustness and to ensure differences between catchments were identified, differences in nutrient use and surpluses between farm types and catchments were also investigated. Farms were classified according to Defra's robust farm type classification system.

6.2.5 Data analysis

Results were analysed using a generalised ANOVA performed by GENSTAT v12. Year (before / after) and mitigation level (GAP / EGAP) were included as factors to test for differences in nutrient use / surpluses before and after mitigation and at different mitigation levels. Farm type and catchment were also included in the treatment structure. ANOVA's were performed on an all interactions basis to explore farm type and catchment specific responses to mitigation. 'Before and after' comparisons were supported by investigations of annual variability across all four years on both a surplus and individual input / output basis. Only those farms providing four years of farm data could be included in the latter (see Table 3-3); corresponding 'before' – 'after' comparisons were made to ensure trends were comparable in both datasets. To investigate the origin of differences in nutrient between farm types, relationships between surpluses and stocking rate (LU ha^{-1}) / cereal area (% farm area) were analysed using generalised linear regression using GENSTAT v.12.

6.3 Results

6.3.1 Effect of mitigation on farmgate budgets

Following the implementation of mitigation, 27 farms (79.4%) saw an improvement in their farmgate surplus. The average surplus decreased by 22.5kg N ha^{-1} from 99.6kg N ha^{-1} to 77.0kg N ha^{-1} representing a 28.5% reduction. Responses ranged from a 195.1kg N ha^{-1} reduction (improvement) to a 110.6kg N ha^{-1} increase (deterioration). However analysis of variance confirmed differences in inputs, outputs and surpluses before vs. after mitigation were largely insignificant (Table

6-5). Only in EMEL were surpluses significantly smaller after mitigation (Figure 6-2). Interactions between farm type, year and catchment were not significant; however differences in nutrient use between farm types and catchments were highly significant justifying further discussion on a farm type – catchment basis. Although change in inputs and outputs were not significant, due to the short timescale over which assessments were made, and the cumulative significance of changes (in EMEL) initial trends and tendencies have still been noted in the discussions which follow.

MSA cattle and sheep farms

Surpluses on cattle and sheep farms in MSA were low, averaging 64.9kg N ha^{-1}, a result of low production intensity characterised by low inputs and low outputs (Figure 6-2 and Figure 6-3). Following the adoption of mitigation average surpluses decreased by 50.60kg N ha^{-1} (65.21%) owing to a 31.8kg N ha^{-1} (35.2%) reduction in fertiliser input and a 27.1kg N ha^{-1} (374.4%) increase in crop export. While the differences before and after mitigation on MSA farms were not significant it is worth noting that fertiliser inputs were consistently larger before mitigation; and under similar cropping in 2007 and 2008, fertiliser inputs fell by 29%. A slight increase in stocking was also observed (Table 6-6), which in accordance with the relationship shown in Figure 6-5 would be expected to increase the surplus.

MSA cereal farms

Average surpluses on MSA cereal farms were slightly higher than those on cattle and sheep farms averaging 65.9kg N ha^{-1} (Figure 6-2). However in contrast to cattle and sheep farms low surpluses stemmed from high total output relative to total input, confirming the efficient use of nutrients in arable systems (Figure 6-3). Total inputs averaged 176.8kg N ha^{-1} with fertiliser representing 72.6% and manure imports 20.9%. While fertiliser use was significantly lower in MSA than EMEL, greater reliance on manure imports meant total inputs and surpluses were both significantly larger in MSA than EMEL (Figure 6-3 and Figure 6-4). Following the implementation of mitigation average surpluses on MSA cereal farms decreased by 14.8kg N ha^{-1} (21.64%) however differences before and after mitigation were not significant (Figure 6-2). Improvements stemmed from the accumulation of small changes in N fixation and crop export; fertiliser and manure imports remained relatively unchanged despite being the focus of mitigation (Figure 6-3).

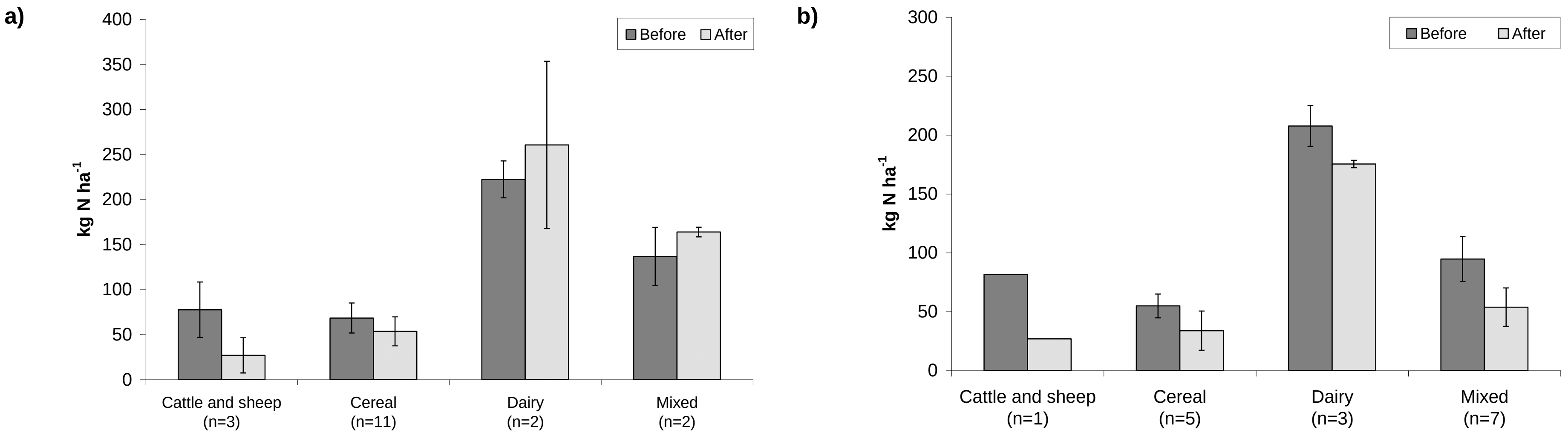


Figure 6-2: Farmgate surpluses (kg N ha^{-1}) before and after mitigation in a) MSA and b) EMEL. Mean and standard error shown with sample size before and after mitigation shown in parentheses. Differences between years significant in EMEL only (see Table 6-5 for full results of statistical analysis).

Table 6-5: Results of farmscale ANOVA analysis in MSA and EMEL. P values refer to the significance of differences between factors for each input / output / surplus.

Catchment	ANOVA factor(S)	INPUTS							OUTPUTS					SURPLUS
		Bedding	Feed	Fertiliser	Manure	Livestock	N Fix.	Total Inputs	Animal Prods.	Crops	Manure	Livestock	Total Outputs	
MSA	Farm type	P<0.001	P<0.001	P<0.001				P<0.001	P<0.001	P<0.001	P<0.001	P<0.001	P<0.001	P<0.001
	Year													
	Farm type x year													
EMEL	Farm type	P<0.001	P<0.001	P<0.001				P<0.001	P<0.001	P<0.001			P<0.001	P<0.001
	Year													P<0.05
	Farm type x year							P<0.1						
MSA+EMEL	Catchment	P<0.001	P<0.001					P<0.05	P<0.001	P<0.05	P<0.05			P<0.05
	Farm type x catch.	P<0.001	P<0.001	P<0.05					P<0.001	P<0.05	P<0.05			
	Farm type x catch. x year													

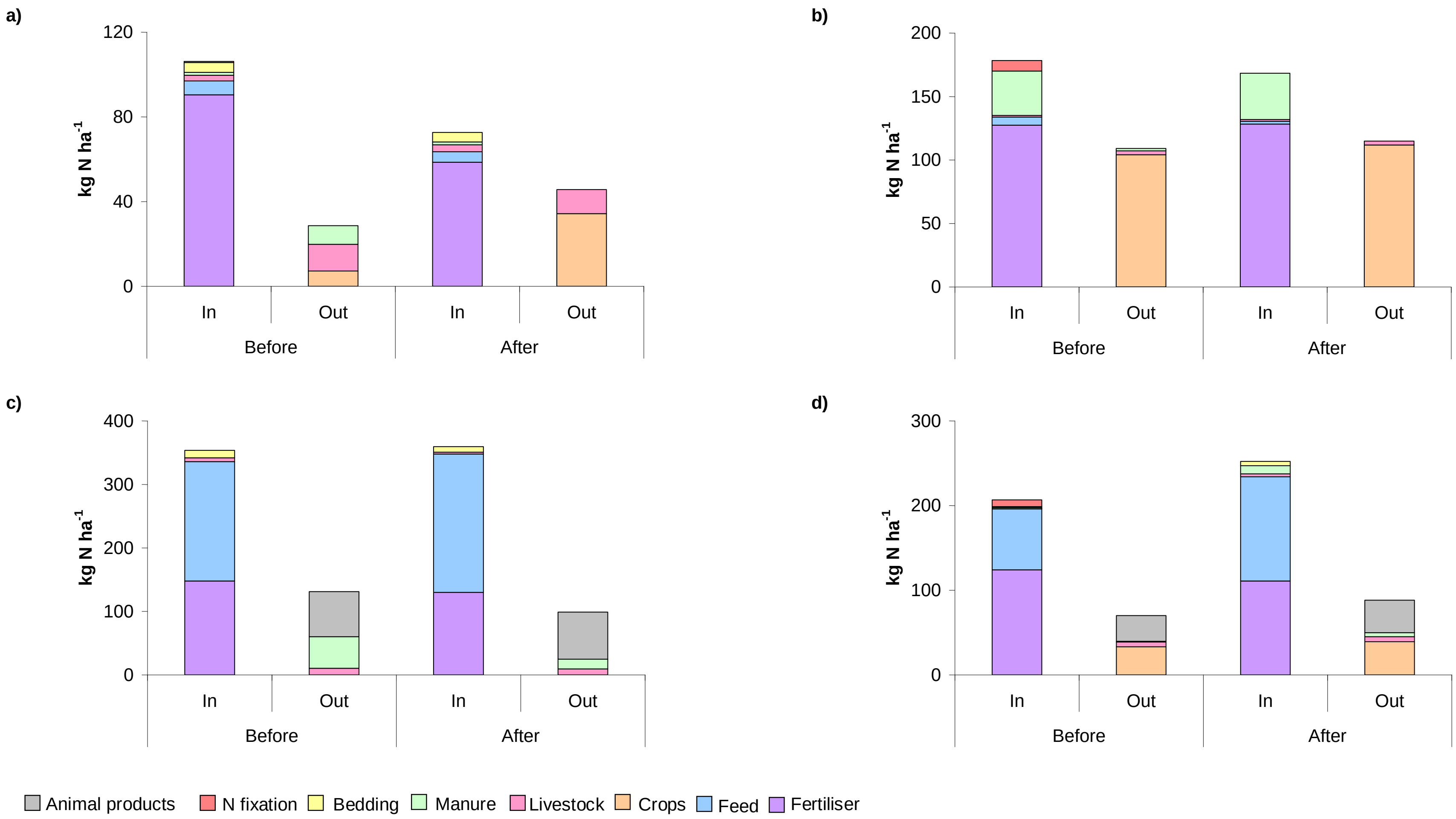


Figure 6-3: Inputs and outputs (kg N ha^{-1}) before and after mitigation on a) cattle and sheep, b) cereal, c) dairy and d) mixed farms in MSA. Mean and standard error shown, with sample size before and after mitigation in parentheses. Note the different scales for each farm type. See Table 6-5 for results of statistical analysis.

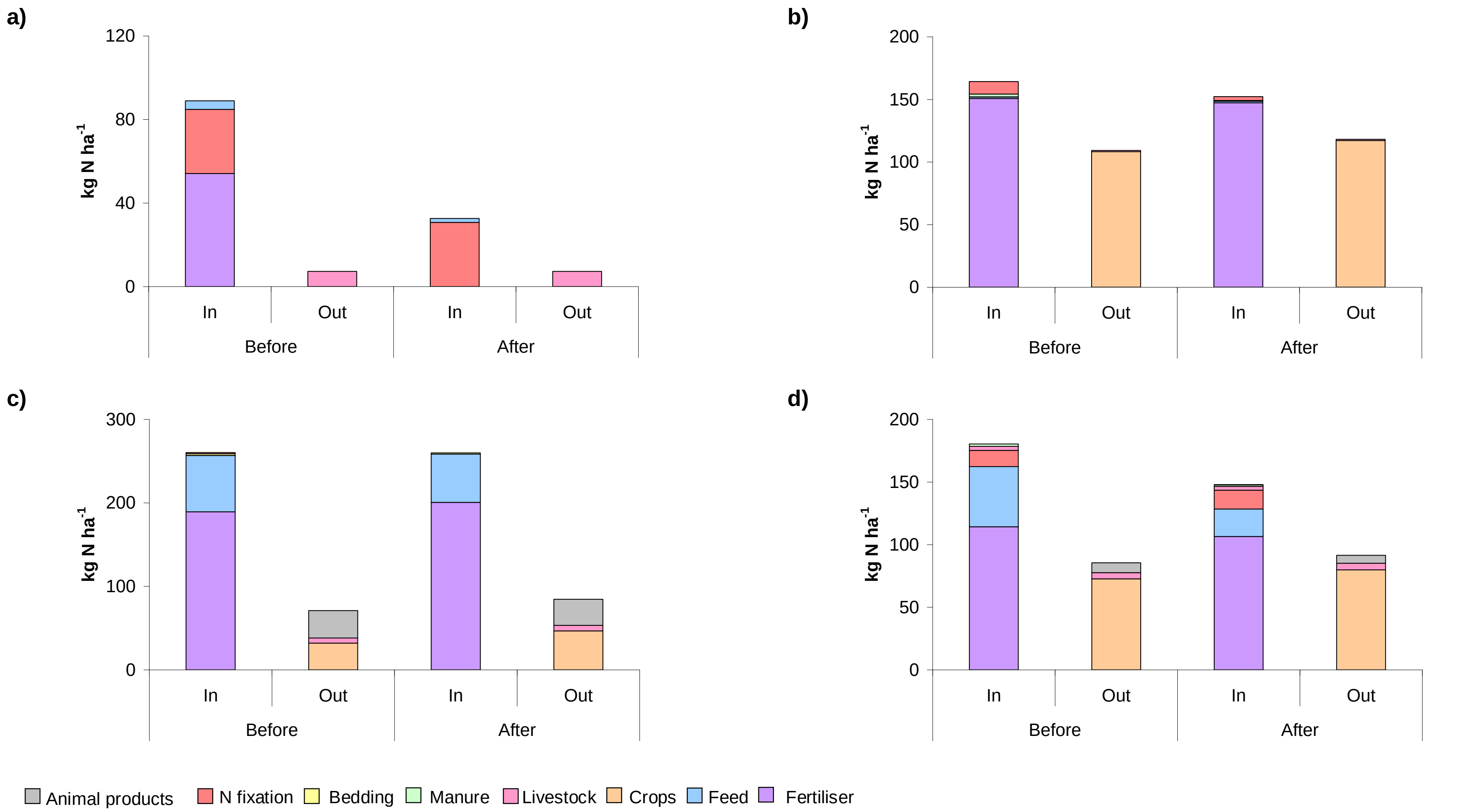


Figure 6-4: Inputs and outputs (kg N ha^{-1}) before and after mitigation on a) cattle and sheep, b) cereal, c) dairy and d) mixed farms in EMEL. Mean and standard error shown, with sample size before and after mitigation in parentheses. Note the different scales for each farm type. See Table 6-5 for results of statistical analysis

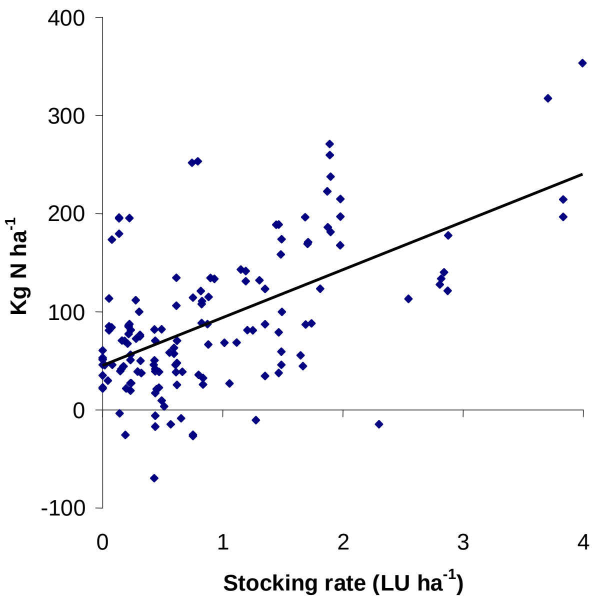


Figure 6-5: Relationship between stocking density (LU ha^{-1}) and farmgate surplus (kg N ha^{-1}) in MSA and EMEL. R^2 = 0.330; P<0.001.

MSA dairy farms

Dairy farms returned the largest surpluses in MSA (232.0kg N ha^{-1}), a result of large feed inputs (Figure 6-2). Higher stocking densities on dairy farms in MSA than EMEL (2.88 vs. 2.05 LU ha^{-1} – see Figure 3-9) meant total inputs were 80.7kg N ha^{-1} (29.4%) larger in MSA leading to significantly larger surpluses on dairy farms in MSA (Figure 6-3 and Figure 6-4). Although differences before and after mitigation were not significant, surpluses on dairy farms were 38.20kg N ha^{-1} (17.18%) larger after mitigation due to a 15.8% (29.7kg N ha^{-1}) increase in feed inputs (Figure 6-2, Figure 6-3). While changes in individual inputs were also non significant, the increase in feed concealed a 17.9kg N ha^{-1} (12.1%) reduction in average fertiliser import and a 34.4kg N ha^{-1} (69.3%) reduction in manure export; the latter suggesting improved utilisation of manure on farm (Figure 6-3). Given that mitigation targeted fertiliser and manure management, manure and fertiliser use may have been affected by mitigation; however, a lack of feed related mitigation meant benefits did not extend to feed inputs or indeed farm surpluses. However it should also be noted that maize production increased at the expense of fertiliser grass in 2008 (after mitigation) (Table 6-6) and the majority of fertiliser reductions occurred prior to mitigation (data not shown).

MSA mixed farms

Mixed farms occupied an intermediate position in MSA reflecting the presence of both inefficient livestock production and efficient arable system. Surpluses on MSA mixed farms averaged 143.5kg N ha^{-1}, almost double that in EMEL (Figure 6-2). This marked difference can be attributed to difference in stocking rates which were almost 50% higher than in MSA than EMEL (1.46 vs. 0.77LU ha^{-1} –Table 6-6); larger inputs were required on MSA farms to support milk production on arable + dairy farms in MSA. This difference supports the relationship observed in Figure 6-5, and highlights the significance of relative arable and livestock enterprises on the nutrient management of mixed farms.

Similar to dairy farms, surpluses on mixed farms increased by 27.2kg N ha^{-1} (19.9%) between pre- and post-mitigation years (Figure 6-2). Although changes in surpluses were not significant, it is worth noting that feed increased by 51.2kg N ha^{-1} (71.3%), counteracting reductions in fertiliser input and increased animal product / crop export (Figure 6-3). However the reduction in fertiliser reflects a steady decline in use across all four years and cannot therefore be linked to the adoption of mitigation.

Table 6-6: Changes in cereal area (% total farm area) and stocking rates (LU ha^{-1}) before and after mitigation in MSA and EMEL. Differences in cereal area and stocking rates between years not significant in either catchment. Differences in cereal area between farm type significant to $p<0.001$ in both MSA and EMEL. Stocking rate significantly different between farm types and catchment to $p<0.001$. A significant farm type x catchment interaction was also observed ($p<0.001$). All other interactions between farm type, year and catchment not significant.

Farm type		MSA			EMEL		
		Base.	Mit.	↑ / ↓	Base.	Mit	↑ / ↓
Cattle and sheep	Cereal	7.5	9.8	↑	0.0	0.0	n/a
	Stocking	1.21	1.24	↑	1.35	1.06	↓
Cereal	Cereal	58.2	70.9	↑	65.9	66.0	↑
	Stocking	0.37	0.30	↓	0.18	0.20	↑
Dairy	Cereal	15.1	19.5	↑	35.9	40.1	↑
	Stocking	2.85	2.99	↑	1.72	1.70	↓
Mixed	Cereal	42.4	50.7	↑	41.3	42.7	↑
	Stocking	1.87	1.60	↓	0.78	0.75	↓

EMEL cattle and sheep farms

Similar to MSA surpluses on cattle and sheep farms in EMEL were low, averaging 68kg N ha^{-1} between 2005-2008 Following the adoption of mitigation surpluses fell by 54.69kg N ha^{-1} (67%) (Figure 6-2), a result of 0kg N ha^{-1} fertiliser inputs post mitigation. However, as observed on MSA cereal farms, the majority of the reduction in fertiliser use (and subsequent to this the reduction in surpluses) occurred between 2006 and 2007 (before mitigation) – data not shown. Although fertiliser and surpluses continued to decrease between 2007 and 2008 a corresponding reduction in the stocking rate may explain this continuing downward trend (Table 6-6).

EMEL cereal farms

Cereal farms returned the lowest surpluses in EMEL averaging 49.7kg N ha^{-1}. Although fertiliser use was significantly higher in EMEL than MSA, low manure imports meant total inputs, and therefore surpluses, were 16.2kg N ha^{-1} (24.6%) lower in EMEL (Figure 6-3 and Figure 6-4). Surpluses on EMEL cereal farms fell by an average of 21.03kg N ha^{-1} (38.29%) post mitigation (Figure 6-2), a result of a reductions in fertiliser and N fixation and an increase in crop export (Figure 6-4). While maximum year on year reductions were observed between 2007 and 2008, and 2008 represented the lowest surplus of all years, annual variability in cropping meant the observed trends might have been occurred irrespective of mitigation. However comparable crop distributions and crop export in 2006 and 2008 exposed lower fertiliser inputs and surpluses post mitigation suggesting improved nutrient management.

EMEL dairy farms

Surpluses on EMEL dairy farms averaged 201.8kg N ha^{-1}, 30.2 kg N ha^{-1} (13.0%) smaller than those in MSA (Figure 6-2); lower stocking rates facilitated lower total inputs whilst total output was supplemented by crop production. A significant improvement in nutrient management was observed on EMEL dairy farms post mitigation with surpluses decreasing by an average of 32.31kg N ha^{-1} (15.55%) (Figure 6-2). Reductions in surpluses stemmed from a 9.4kg N ha^{-1} (13.9%) reduction in feed imports and 14.7kg N ha^{-1}(45.8%) increase in crop export (Figure 6-4). However, surpluses followed a decreasing trend across all years and changes in feed and fertiliser inputs revealed little correspondence with the implementation of mitigation, or indeed changes in farm structure. For example increased feed was not consistent with the progressive reduction in stocking also observed, and nor were changes in feed consistent with changes in the fertilised grass area. Connections could however be made between crop export and arable area; high export in 2006 and 2008 corresponded with larger arable areas, although favourable growing conditions are arguably more influential on crop export. Inconsistencies between nutrient use and farm structure (cropping and livestock numbers) may reflect interactions between years supporting the use of longer term averages both before and after mitigation.

EMEL mixed farms

Similar to MSA, surpluses on mixed farms in EMEL occupied an intermediate position, however on average surpluses were almost half those in MSA (84.6 vs. 143.5kg N ha^{-1}) (Figure 6-2). Arable production represented a much larger component of total production on EMEL mixed farms resulting in considerably lower total inputs.

In contrast to MSA, mixed farms in EMEL returned significantly lower surpluses after mitigation (Figure 6-2), which stemmed from a 26.2kg N ha^{-1} (54.4%) reduction in feed, 7.8kg N ha^{-1} (6.8%) reduction in fertiliser inputs, and 7.2kg N ha^{-1} (9.9%) increase in crop export (Figure 6-4). However a downward trend in surpluses and feed imports across all years suggests improvements were not a result of mitigation. Furthermore, WAgriCo mitigation targeted fertiliser and manure management and thus was unlikely to result in substantial reductions in feed imports. Changes in nutrient use could not however be linked to changes in cropping and livestock numbers either, highlighting the complex interactions between arable and livestock production and the difficulty in attributing change to mitigation on mixed farms. Reduced feed and fertiliser inputs may reflect an attempt to improve nutrient utilisation prior to WAgriCo or advanced purchasing of inputs to buffer increased feed / fertiliser price. Moving averages or a longer study would be required to account for changes in farm stocks more explicitly.

6.3.2 Sensitivity to mitigation level (GAP vs. EGAP) and manure management plans

Mitigation codes (GAP vs. GAP+EGAP and +MMP vs. –MMP) had no significant effect on farm surpluses prior to mitigation (see section 3.5.1 for details of codes) suggesting prior awareness of effective nutrient management / environmental issues had minimal influence on the level of mitigation adopted. On a before vs. after basis, larger improvements appear to correspond with higher levels of mitigation (Figure 6-6) however ANOVAs confirmed this observation to be insignificant.

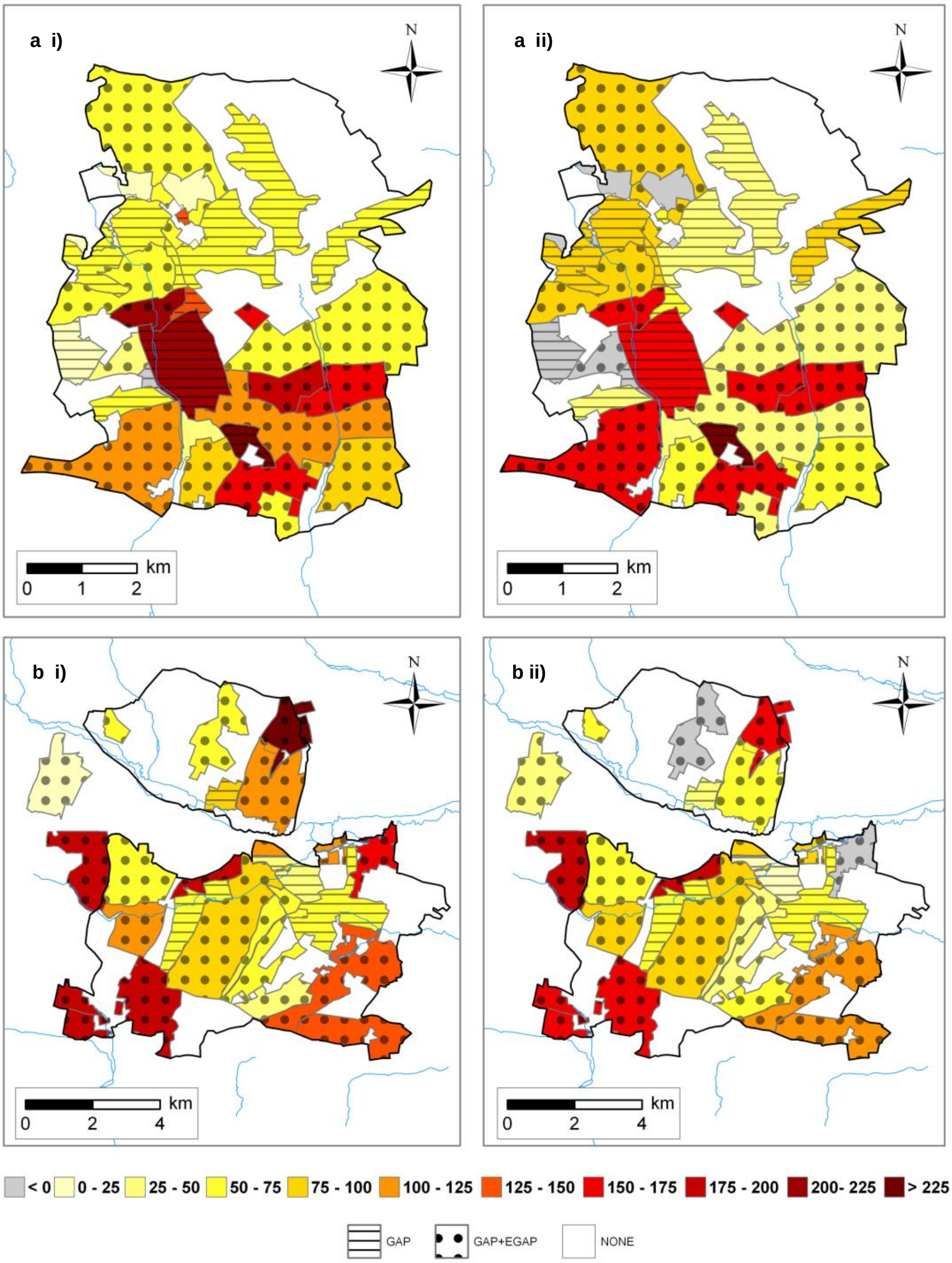


Figure 6-6: Effect of mitigation code on farmgate surpluses (kg N ha^{-1}) before (i) and after (ii) mitigation in a) MSA and b) EMEL.

6.4 Development

6.4.1 Mitigation scenarios

6.4.1.1 Introduction and methodology

To broaden the range of mitigation methods evaluated and to increase the degree of change mitigation induced, a series of farm scale mitigation scenarios were simulated. In accordance with Cherry et al., (2008) only those methods to which budgets were likely to show a degree of sensitivity to were considered. From the options available, selection was predominately based on their suitability to farm scale simulation; some methods are self compensating over the whole farm, and others alter only the distribution of surpluses between fields and not the overall farm surplus better suiting them to field scale analysis. Applicability to a wide range of farms and ease / certainty of the simulations were also considered. Those methods likely to induce uncertain compensatory behaviour for example were avoided. (Full details of the selection process can be found in table A-3 in the appendix). Table 6-7 details those methods selected for scenario analysis, the simulation approach adopted, and reasons for inclusion.

Corresponding with field scale scenario analysis, mitigation scenarios were applied to MSA farms only, with budgets adjusted and recalculated to reflect the simulated mitigation. Due to simulation complexity and lower applicability, 'case study' simulations were conducted for farms where impact was likely to be greatest (maximum dependence on imported feed, % farm area in arable production, stocking rate for case studies 1-3 respectively). Case study simulations were performed in all four years, the adjustment of 'mitigation' results being justified on the grounds that simulated change would be much greater than that attributed to the observed implemented mitigation. In both strands of work modelled results were (also) compared to 'baseline' results; however, adopting a suitable 'baseline' for comparison proved more complex than in similar field scale work. Whilst a comparison with 2008 results (Mitigation 1, Table 6-7) would provide comparable cropping and livestock situations, results are already affected by the mitigation. However by adopting 2005-2007 results as a 'baseline' ('baseline' 2, Table 6-7), output and actual 2008 results would reflect different cropping and livestock situations, which, as section 6.3.1 highlighted, would potentially induce differences in nutrient management irrespective of simulated mitigation. A hybrid 'baseline' figure was therefore developed ('baseline' 1, Table 6-7), applying 'baseline' fertiliser applications to 2008 cropping and retaining all other 2008 farm data. Although the direct link between fertiliser application and yield is lost ('baseline' fertiliser is combined with 2008 output), yield maximising applications are likely to have been made across all years. By retaining 2008 livestock data (but not fertiliser data) it is assumed that little change was made to feed management (unlike fertiliser management) as a result of mitigation. Although large changes in feed were observed on some farms, this was not a direct result of WAgriCo mitigation.

Table 6-7: Details of farm scale mitigation scenarios

Name	Description	Justification	Simulation Approach
'Baseline' 1	*Hybrid 'Baseline'*	*See text below*	*'Baseline' fertiliser projected onto 2008 cropping. All remaining data from 2008.*
'Baseline' 2	*Actual 'Baseline' results*	*See text below*	*Average of observed 2005 – 2007 results*
Mitigation 1	*Actual 2008 'mitigation' results*	*See text below*	*Observed 2008 results*
Mitigation 2	Fertiliser recommendation (inc. planned manure applications)	– Wide applicability – Availability of fertiliser recommendations[1]	Fertiliser input calculated using fertiliser recommendations, taking account of planned manure applications and applied to 2008 cropping. All remaining data from 2008.
Mitigation 3	Fertiliser recommendation (exc. Planned manure applications)	As Mitigation 2 plus… – To investigate the impact of replacing fertiliser with manure.	Fertiliser input calculated using fertiliser recommendations NOT taking account planned manure applications, and applied to 2008 cropping. All remaining data from 2008.
Mitigation 4	Integration of manure N (based on readily available N (RAN) and fertiliser recommendation)	– To address the absence of explicit manure accounting in farmgate budgets and their lack of sensitivity to manure based mitigation. – To investigate scope for improved manure management which is typically high.	Fertiliser input calculated using fertiliser recommendations (not taking account of planned manure) less the total farm RAN. RAN calculated from net farm manure N (Excreta + imported manure – exported manure) using the appropriate RAN factor (0.1, 0.25 and 0.5 for old FYM, fresh FYM, slurry / layer manure - values from MANNER (Chambers et al., 1999)). All remaining data from 2008.
Mitigation 5	Integration of manure N (based on RAN and 'baseline' fertiliser[2])	As Mitigation 4 plus… – Fertiliser recommendations considered too high by WAgriCo farmers.	Fertiliser input calculated using 'baseline' fertiliser applied to 2008 cropping less the total farm RAN (see above). All remaining data from 2008.
Case study 1	Reduction of feed N	– Feed related mitigation best explored at the farm scale – field scale impact more indirect	In accordance with Cuttle et al., (2006) crude protein (CP) content of concentrates reduced to 14%.

Name	Description	Justification	Simulation Approach
Case study 2	Arable reversion	– Field scale surplus effectively zero therefore better suited to farm scale investigation.	20% of arable area converted to grassland (based on 14% uptake of premium grassland option in Nitrate Sensitive Area Scheme in which arable land was converted to zero or low input grassland (Lord et al., 1999). Arable fertiliser and arable output reduced by 20%. Manure to arable land reduced by 20% and added to farm export.
Case study 3	Reduction in stocking density[3]	– Farm scale simulation preferable to avoid issues related to distribution of grazing and excreta/ urine, and time spent in housing.	After Fezzi et al., (2008) and Cuttle et al., (2004) livestock density reduced by 20%. Feed, bedding, animals in, animals out, animal products out and manure out reduced by 20%. Imported manure increased by 20%, fertiliser to grass reduced by 20%, and 20% less arable produce retained. Changes to fodder production and arable crop areas kept constant for simplicity.

[1] Fertiliser recommendations were provided during the WAgriCo project, however farms could be recognised as adopting GAP level mitigation without implementing them so long as a reason was offered.

[2] Risk of double counting where manure N is already accounted for in fertiliser applications. However the extent to which fertiliser recommendations exceeded actual applications is likely to far exceed the degree of double counting especially when utilising pre-mitigation data.

[3] For maximum impact a whole farm reduction was simulated meaning no corresponding reduction in grass area.

6.4.1.2 Results

'All farm' mitigation scenarios

Across all farm types the integration of total farm readily available manure (RAN) within 'baseline' fertiliser supply (scenario 5) was most effective in reducing farmgate surpluses (Figure 6-7 and Figure 6-8). Where 'baseline' fertiliser was replaced with fertiliser recommendations impact was less positive, with improvements no longer observed on cattle and sheep farms. Fertiliser recommendations scenarios (scenarios 2 and 3) were least effective, inducing larger surpluses on all but dairy farms, confirming recommended applications were higher than observed rates. The inclusion of planned manure applications within scenario 2 meant surpluses were consistently lower / equal to those of scenario 3. Differences between the two expose a higher level of manure accounting on cattle and sheep and cereal farms. Consequently the risk of double counting of manure N within scenario 5 is higher for these farms. Differences in 'baseline' surpluses meant the performance of each scenario was farm type specific. Further comment is therefore offered on a farm type basis.

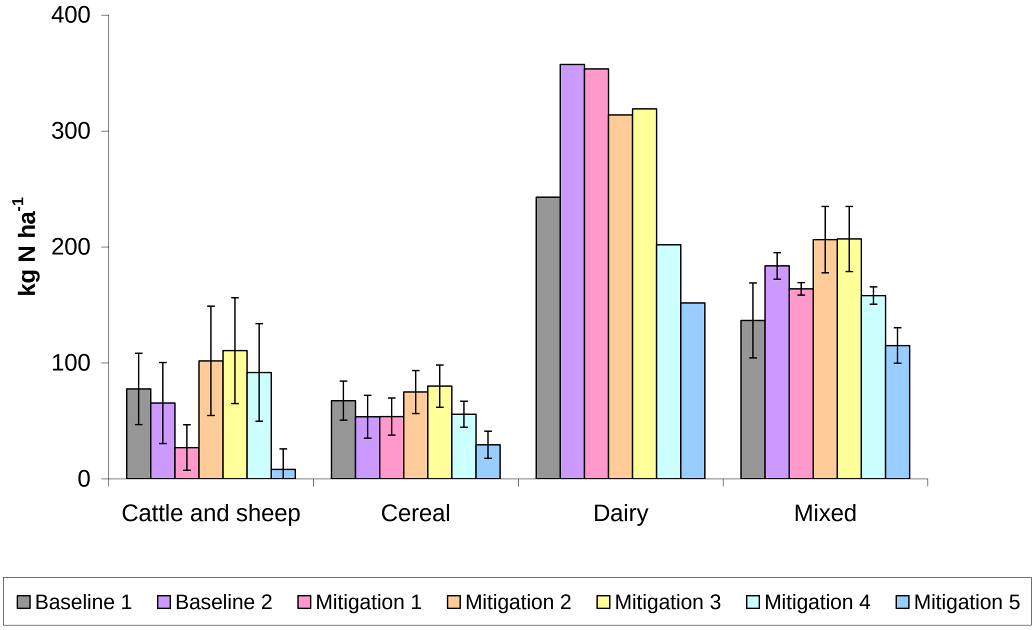


Figure 6-7: Modelled effect of mitigation scenarios on MSA farm surpluses. Figure shows farm type average surplus (kg N ha^{-1}) under baseline and after mitigation scenarios – see Table 6-7 for full details of each scenario. Error bars show standard error. Missing fertiliser recommendations meant only one dairy farm could be included in scenario analysis; as a result error bars are not present.

On cattle and sheep farms only scenario 5 had a positive impact, however given the large relative change, improvements were the largest of any farm type specific scenario performance (Figure 6-8). In contrast, scenarios involving fertiliser recommendations had the largest negative impact, highlighting the extent to which fertiliser recommendations exceeded observed applications on grass, the predominate land use on cattle and sheep farms. A similar picture was observed on cereal farms, albeit to a smaller degree. Compliance with fertiliser recommendations again increased surpluses, and given the low ‘baseline’ surpluses resulted in large % change. Scenario 5 was again most effective, however when compared against the observed ‘baseline’ surplus, scenario 4 was also beneficial despite utilising fertiliser recommendations. While comparisons with observed ‘baseline’s’ are questionable given the differences in cropping and livestock situations, the result highlights the extent to which integrating RAN within fertiliser supply can reduce surpluses even on predominately arable farms. It is perhaps useful to note that the average farm RAN was approximately three times higher on cereal farms than cattle and sheep farms, the latter being predominately small, extensive farms.

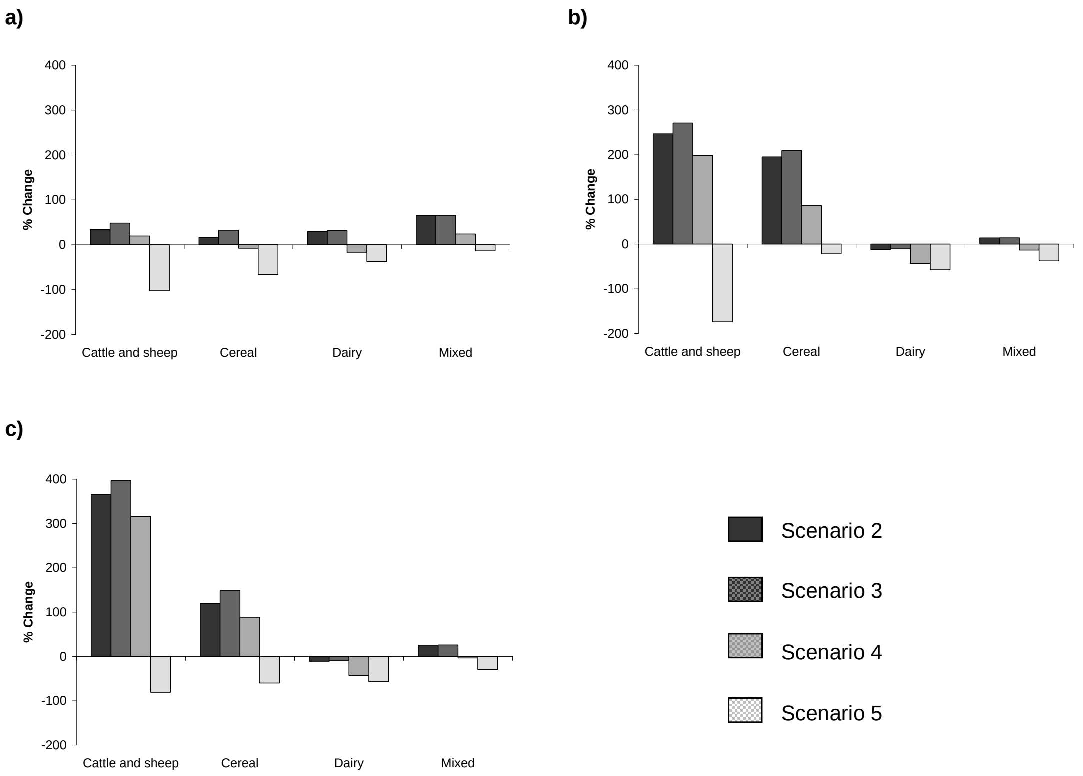


Figure 6-8: Effect of mitigation scenarios on modelled MSA farm surpluses shown on a % change from a) observed 2005-2007 average (baseline 1) b) hybrid ‘baseline’ (baseline 2) c) observed 2008 post mitigation average (mitigation 1). See Table 6-7. for details of ‘baseline’ and mitigation scenarios.

Unlike all other farm types, the implementation of fertiliser recommendations on dairy farms (scenarios 2 and 3) represented a positive change when compared with the hybrid ‘baseline’ (Figure 6-8b). While improvements did not extend to comparisons with the observed ‘baseline’, comparable cropping and livestock situations between hybrid ‘baseline’ and scenario output meant this result was more indicative of positive impact than comparisons with the observed ‘baseline’. Both RAN scenarios effectively reduced dairy farm surpluses, the difference between the two scenarios less than other farm types because of the positive impact of fertiliser recommendations. A slightly different situation was observed on mixed farms with fertiliser recommendations again detrimental to surpluses, albeit to a smaller degree than on cattle and sheep and cereal farms. Whilst scenario 5 was beneficial against both ‘baseline’s, responses to scenario 4 were mixed, with positive impact restricted to comparisons with the hybrid ‘baseline’ (Figure 6-8b). However, as mentioned above this is more suggestive of a robust improvement than where improvements are seen only in Figure 6-8a. Differences between the 2 scenarios were again relatively small owing to the small difference between recommended and observed fertiliser applications.

Table 6-8: Results of 'case study' mitigation scenarios

Mitigation simulation	2005	2006	2007	2008
Case study 1 - Reduction in feed N				
Observed (kg N ha^{-1})	196.75	214.36	317.55	353.47
Simulated (kg N ha^{-1})	184.41	202.01	301.1	284.01
Reduction (%)	6.27	5.76	5.18	19.65
Case study 2 - Arable reversion				
Observed (kg N ha^{-1})	195.97	179.46	195.08	173.53
Simulated (kg N ha^{-1})	154.36	141.15	158.64	136.41
Reduction (%)	21.23	21.35	18.68	21.39
Case study 3 - Reduction in stocking density				
Observed (kg N ha^{-1})	177.85	140.3	188.8	158.51
Simulated (kg N ha^{-1})	143.18	108.07	154.50	125.84
Reduction (%)	19.50	22.97	18.17	20.61

Due to low feed imports in 2005 and 2006, and large total inputs in 2007, a reduction in feed N had little impact on farmgate surplus in the first 3 years (Table 6-8). However in 2008 the relative increase in feed exceeded that of total inputs resulting in a larger reduction in the overall surplus. An increase in the stocking density was also witnessed over the 4 years, however this does not explain the increase in feed, or the increased mitigation impact in 2008. Arable reversion revealed a more consistent improvement, averaging 20.7% over the 4 years; reduced variability in 'baseline' surpluses in part explaining the lower variability in scenario output. Impact was however slightly lower in 2007 when higher arable produce retention increased the apparent reduction in crop export (retained crop export was kept constant regardless of the total farm produce), and 'baseline' surpluses were at their largest. Whilst the surplus reduction was similar to the arable area foregone, should the arable enterprise represent a smaller fraction of the farm, the observed improvement would increasingly deviate from the area converted. It should also be noted that the 'baseline' surplus was unusually high for a cereal farm, owing to large layer manure import, however the relative impact of arable reversion would not decrease should it be extended to low surplus cereal farms.

Improvements following a reduction in stocking density were of a similar magnitude to those observed under arable reversion. Maximum impact was observed in 2006 when 'baseline' surpluses were low. Although feed and fertiliser inputs fluctuated between years, the relative degree of change was more influential than absolute adjustments. The level of improvement was again similar to the magnitude of change induced.

Table 6-9: Comparison of farm scale mitigation scenarios including case studies. Results shown for specific case study farms and for corresponding farm type average (in blue). Note that case study results are for 2008 only, and in the case of dairy farms included only one farm meaning results are identical at individual farm and farm type level. Negative % change values denote an improvement in the farmgate surplus.

	Case study	Scenario 2		Scenario 3		Scenario 4		Scenario 5	
Dairy farm - reduced feed									
Surplus (kg N ha^{-1})	**284.01**	313.91	313.91	319.06	319.06	201.89	201.89	151.76	151.76
Change from Base. 2 (%)	***-20.52***	*-12.16*	*-12.16*	*-10.72*	*-10.72*	*-43.50*	*-43.50*	*-57.53*	*-57.53*
Change from Mit.1 (%)	***-19.65***	*-11.19*	*-11.19*	*-9.74*	*-9.74*	*-42.88*	*-42.88*	*-57.07*	*-57.07*
Cereal farm - arable reversion									
Surplus (kg N ha^{-1})	**136.41**	235.32	74.87	235.32	79.94	139.01	55.67	76.47	29.45
Change from Base. 2 (%)	***-26.27***	*27.19*	*194.97*	*27.19*	*208.86*	*-24.86*	*85.84*	*-58.67*	*-21.71*
Change from Mit.1 (%)	***-21.05***	*36.19*	*119.26*	*36.19*	*148.26*	*-19.55*	*88.13*	*-55.74*	*-60.13*
Mixed farm - reduced stocking density									
Surplus (kg N ha^{-1})	**125.84**	177.62	206.31	178.84	206.92	150.58	158.02	130.24	114.98
Change from Base. 2 (%)	***-35.52***	*-8.98*	*13.75*	*-8.35*	*14.06*	*-22.84*	*-13.37*	*-33.26*	*-37.67*
Change from Mit.1 (%)	***-20.61***	*12.06*	*25.46*	*12.83*	*25.84*	*-5.00*	*-3.61*	*-17.83*	*-29.46*

The improvements observed on case study farms were comparable, and in some cases greater than those observed under the mitigation scenarios applied to all MSA farms (Table 6-9). Improvements from hybrid 'baseline's' were generally larger than those from actual 2008 results, an observation not surprising given the reductions in surpluses observed between 'baseline' and mitigation years on 76% of farms. Reductions in feed N represented an intermediate approach, performing better than fertiliser recommendations (scenarios 2 and 3) but less effectively than manure based mitigation (scenarios 4 and 5); the reduction in feed N being considerably smaller than unaccounted RAN, but larger than the difference between actual and recommended fertiliser. Arable reversion and a reduction in stocking density proved more effective than feed N reductions with improvements consistently exceeding those of scenarios 2, 3 and 4; high fertiliser recommendations reduced the effectiveness of scenario 4 on cereal and mixed farms. Across all cereal farms arable reversion was also more effective than scenario 5, however on the case study farm, high RAN and lower fertiliser recommendations meant this trend was reversed. On mixed farms the opposite was observed with high RAN on non case study farms improving the performance of catchment wide scenarios relative to the case study farm only.

Hybrid 'baseline' – exposing the effect of cropping on surpluses

Comparison of observed 'baseline', hybrid 'baseline' and observed mitigation surpluses also provided an opportunity to isolate changes in cropping and livestock from that of nutrient management on MSA farms. Differences in actual and hybrid 'baseline' surpluses arise from differences in cropping areas and general farm inputs and outputs. Differences between the hybrid 'baseline' and observed mitigation reveal differences in fertiliser and manure applications, and are comparable to the comparisons made at field scale between 'baseline' and mitigation results. On both cattle and sheep and cereal farms hybrid surpluses are lower than those observed in 2005-2007 meaning crop and livestock situations in 2008 were likely to yield lower surpluses than previous years (Figure 6-7). On cattle and sheep farms a further reduction was observed between the hybrid 'baseline' and the observed mitigation surplus, indicative of changes in nutrient management. Differences between hybrid and mitigation surpluses were however very slight on cereal farms suggesting minimal mitigation driven change. On dairy and mixed farms both hybrid 'baseline' and observed mitigation surpluses were considerably larger than those observed between 2005 and 2007, and a small reduction was observed between hybrid and actual mitigation results (Figure 6-7). Cropping and farm inputs and outputs were therefore likely to increase surpluses considerably in 2008, as was observed. However the hybrid surplus suggests a small reduction in fertiliser and manure applications was still observed.

6.4.2 Upscaling farm surpluses to the catchment scale

6.4.2.1 Introduction and methodology

To maintain legislative relevance and facilitate inter catchment comparison, farm gate methodologies were extended to the catchment scale. In keeping with chapter 5, catchment surpluses were derived using an 'enlarged' farm methodology; farm and catchment methodologies were therefore conceptually similar. Essentially treating each catchment as a large farm, total inputs and total outputs on individual farms were summed net of any internal transfers. Net catchment outputs were then subtracted from net catchment inputs, and presented on a 'per ha' basis. In doing so the balance considers farm size, calculating an area weighted contribution for each farm, similar to Domburg et al., (2000) and Daalgard et al., (2002). However, while the frequency distribution of farm types is reflected in output, given the effect of farm type on nutrient use, results should be interpreted in conjunction with this information.

6.4.2.2 Results

Fertiliser dominated nutrient inputs in both catchments across all years, representing 64 – 69% of the N input to MSA and 73 – 82% in EMEL (Figure 6-9). Larger manure inputs of up to 15% compensated for lower fertiliser inputs to MSA and reflect lower stocking densities in MSA (Figure 6-10). Reliance on N fixation (through leguminous crops / clover rich grass) was small in MSA but averaged 8% in EMEL. Feed inputs were slightly higher in MSA than EMEL (16% vs. 13%) corresponding with proportionally greater milk production in MSA. In terms of outputs, crop export was the dominant flux equating to 82 and 86% of total outputs in MSA and EMEL respectively. However in line with feed imports milk was a larger source of output in MSA than EMEL. Livestock export was however minimal in both catchments, equating to just 5% of total output. Fertiliser input and crop export was proportionally larger than that recorded in a Scottish study by Domburg et al., (2000). Fertiliser represented 60% of inputs and crop export 64% of output in the latter. Livestock export was however much larger from the Scottish catchment, equating to 32% of total output. Differences in output reflect the prevalence of livestock farms in the Scottish catchment and the predominately arable land use in MSA/EMEL. Higher N fixation and lower production intensity explain the reduced dependency on fertiliser in the Scottish catchment.

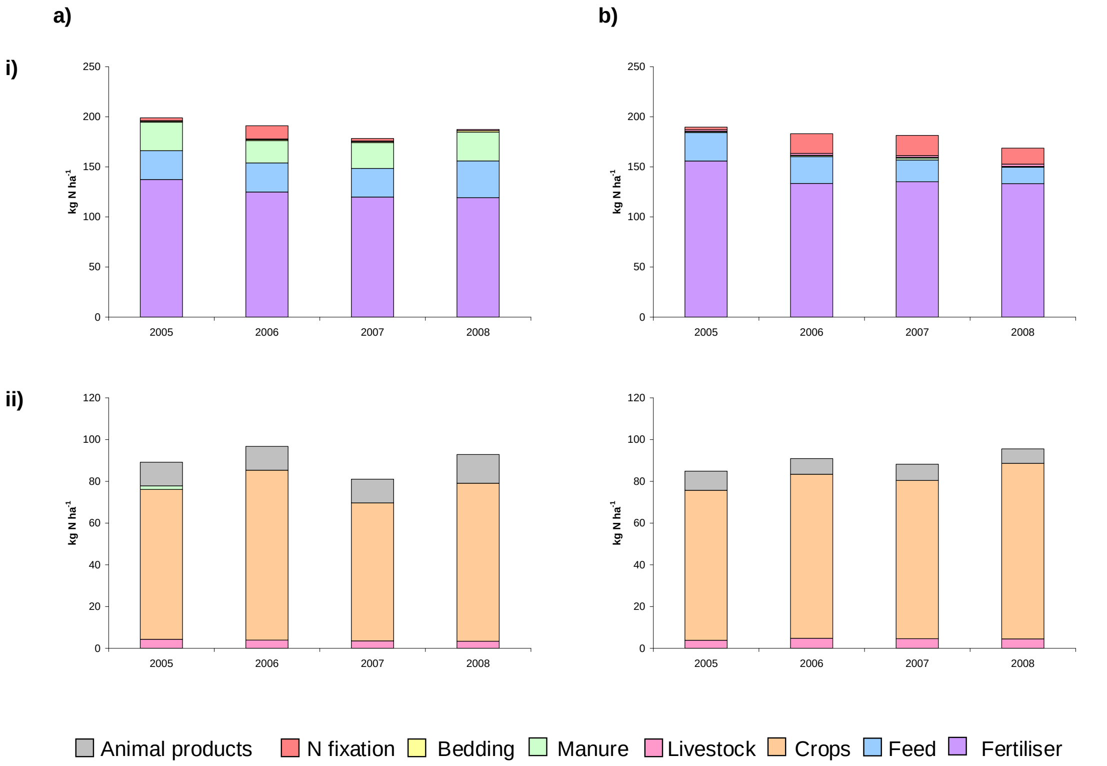


Figure 6-9: Annual catchment i) inputs (kg N ha^{-1}) and ii) outputs (kg N ha^{-1}) in a) MSA and b) EMEL

In both MSA and EMEL, low crop export coupled with high fertiliser inputs (Figure 6-9) meant surpluses were larger in 2005 than other years (Table 6-11). In 2006 MSA and EMEL surpluses were similar, but a low 2008 surplus in EMEL meant only the MSA 2006 surplus appeared small. However a 9 / 14% reduction in fertiliser and 13 / 10% increase in crop export (MSA / EMEL respectively) meant surpluses were considerably smaller in 2006 than 2005 in both catchments. 2007 saw a further reduction in total inputs in both catchments, but was accompanied by lower crop export; crop export fell by nearly 20% in MSA (Figure 6-9). As a result surpluses increased in both catchments (Table 6-11). In 2008 nutrient use in the two catchments differed widely. Maximum output and minimum input (of the four years) meant the EMEL surplus was 21kg N ha^{-1} lower than that in MSA. Although fertiliser inputs fell slightly, the reduction in inputs observed in EMEL stemmed predominately from a reduction in feed (25%) and N fixation (21%). In MSA crop export fell short of its 2006 peak and returned higher total inputs than 2007 owing to a 28% increase in feed (Figure 6-9).

Table 6-10: Comparison of catchment and farm scale surpluses (kg N ha^{-1}). Catchment surpluses pre-mitigation shown in italics, and change between pre- and post-mitigation shown in brackets.

Catchment	Scale	2005	2006	2007	2008
MSA	*Catchment*	109.83	94.33	97.34	94.56
			100.47		(-5.88%)
	Farm	104.66	97.16	102.25	92.75
EMEL	*Catchment*	104.76	92.23	93.24	73.16
			96.74		(-24.37%)
	Farm	107.55	102.09	92.95	71.36

Surpluses decreased between pre- and post-mitigation years in both catchments, however annual variability means improvements cannot be directly attributed to mitigation. Differences in crop distribution explain a significant proportion of this variability, especially in MSA. Large grass and WOSR (MSA only) areas account for the high fertiliser inputs to both catchments in 2005 (Figure 6-10), and on farm consumption of grass explains the low crop export. The large surpluses recorded this year elevated pre-mitigation surpluses and demonstrate that improvements between 2007 and 2008 (corresponding with the implementation of mitigations) are not necessary to observe improvements post- mitigation. Large areas of WW (MSA) / arable areas (EMEL) corresponded with high crop export in 2006 and 2008, while in 2007 large areas of oats with a low input – output balance reduced crop export in MSA (Figure 6-10). In both catchments correspondence between maximum crop export and minimum surpluses highlights the significance of cropping decisions and environmental factors on catchment surpluses (Figure 6-9 and Table 6-11). While maximum reductions in fertiliser were observed between 2005 and 2006, inputs were at their lowest in 2008 in both catchments (Figure 6-9). However the downward trend across all years suggests reductions were not driven solely by mitigation. Despite stocking densities remaining relatively unchanged throughout the study (Figure 6-10), feed and manure imports to MSA increased in 2008; the latter counteracting reductions in fertiliser. Given the changes in cropping and increased nutrient use in 2008, mitigation appears to have had little impact in MSA. Reductions in feed and fertiliser, the latter not fully explained by changes in cropping, coupled with a maximum surplus reduction between 2007 and 2008 mean EMEL results indicate a positive response to mitigation.

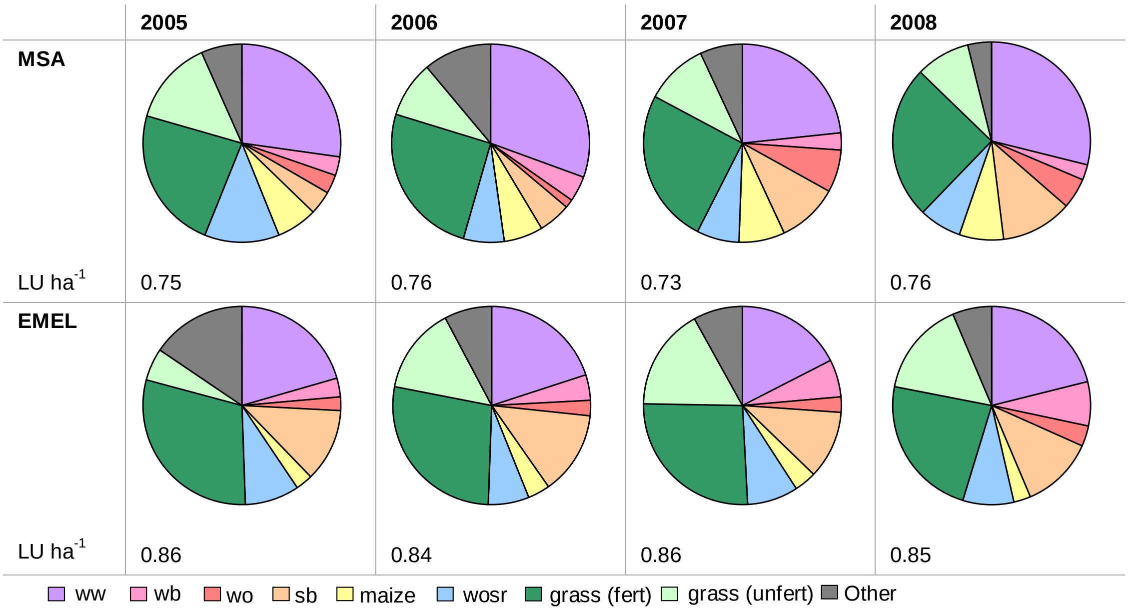


Figure 6-10: Annual catchment average crop distribution and stocking density (LU ha^{-1}) in MSA and EMEL.

6.4.2.3 Correspondence between farm and catchment scale results

A statistically significant difference between farm scale pre- and post- mitigation results and a general downward trend in farm type average surpluses between 2005 and 2008 meant EMEL farm scale results corresponded well with the progressive reduction observed at the catchment scale (Table 6-11). In contrast MSA responses were farm type specific, exaggerated by differences in farm size (see Table 3-3): thus, farm and catchment results differed (Table 6-11). Whilst 2005 represented the largest catchment surplus by more than 11kg N ha^{-1}, only on cereal farms were large surpluses observed (Table 6-6). However the largest of these surpluses occurred on farms exceeding 200ha resulting in large area weighted contributions. Although the 2008 catchment surplus was not the lowest of the four years, its relatively small size contradicts the maximum surplus observed on dairy and mixed farms (Table 6-6). It should be noted that the MSA dataset contains only two dairy and two mixed farms, one of which is only 31ha. Whilst surpluses may have increased at the farm scale, impact at the catchment scale is therefore less significant.

6.4.2.4 Up-scaling mitigation scenarios to the catchment scale

Given the significance of farm type distribution and farm size on catchment surpluses, and the legislative relevance of catchment scale assessment, mitigation scenario simulations were also upscaled using the methodology described in section 6.4.2.1. With mitigation scenarios simulated only on MSA farms, catchment results were also restricted to MSA. In agreement with farm scale results, the integration of RAN within fertiliser supply (mitigation scenarios 4 and 5) proved most effective at the catchment scale, with a surplus reduction from the hybrid 'baseline' of 43.54% observed under scenario 5 (Figure 6-11). Improvements under scenario 4 were more modest with the catchment surplus falling by 13.65%; the inclusion of high fertiliser recommendations reducing the impact of RAN integration. Scenarios 2 and 3 both increased the surplus, highlighting the widespread over estimation of fertiliser requirements in fertiliser recommendations. Figure 6-12 provides further evidence that scenarios 4 and 5 outperform scenarios 2 and 3.

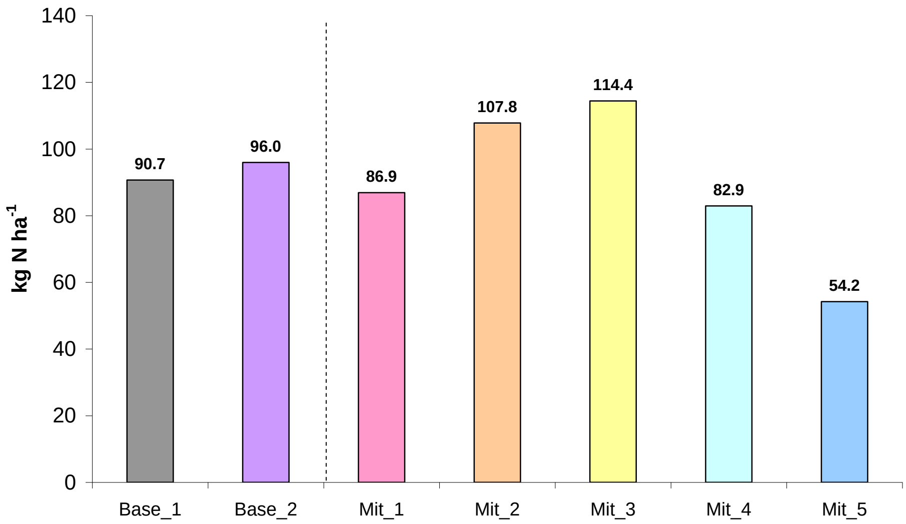


Figure 6-11: Effect of mitigation scenarios on MSA catchment surplus (kg N ha^{-1}). Due to incomplete fertiliser recommendations, 'baseline' and mitigation results presented here differ slightly from those in Table 6-11. See Table 6-7 for full details of mitigation scenarios.

Catchment results generally corresponded with those observed at the farm scale, the success of scenario 5 at the catchment scale reflecting the consistently positive response observed at the farm scale. Catchment results did however conceal farm type specific responses, for example fertiliser recommendations had a positive impact on dairy farms despite a 19.7% increase in the catchment surplus (scenario 3 – see Figure 6-8). Further, whilst scenario 4 had a positive impact at the catchment scale, relatively low RAN and high fertiliser recommendations meant introduction on cattle and sheep and cereal farms had a negative impact. Differences in farm type responses combined with area weighted contributions meant the magnitude of change observed at the catchment scale was smaller than that at the farm scale. In particular the large % improvements observed on small farms, a result of low 'baseline' surpluses, had less effect on catchment than farm scale results.

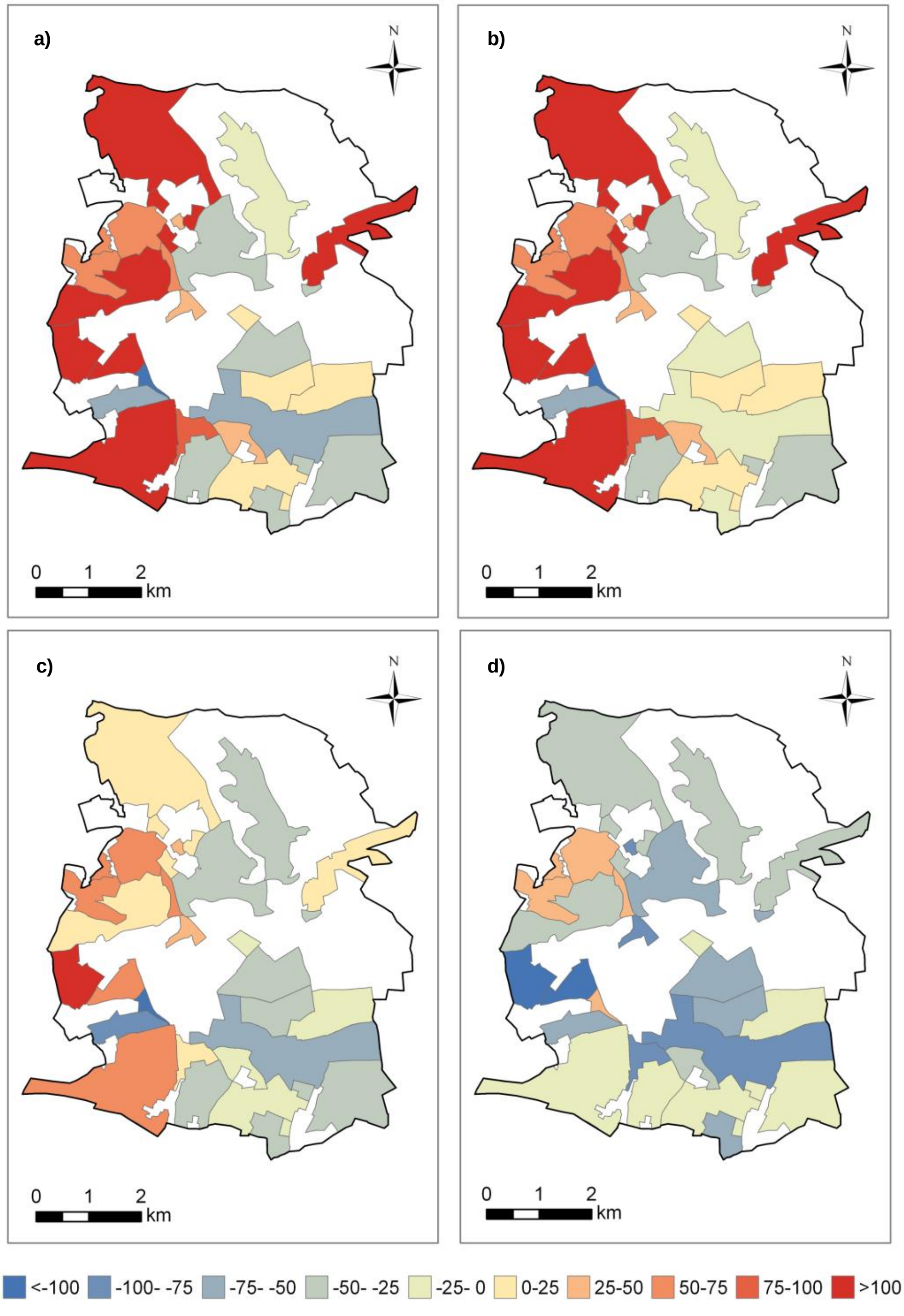


Figure 6-12: Effect of mitigation scenarios on farmgate surpluses - % change from 'baseline'. Blue = reduction in surplus = improvement. Red = increase in surplus = deterioration.

6.5 Discussion

6.5.1 Sensitivity of farmgate budgets to mitigation

Following the implementation of mitigation, farm gate surpluses improved on 79% of farms; this is in agreement with the 70% of Swedish dairy farms which reduced surplus following an environmental awareness campaign aimed at improving manure utilisation (Swensson, 2003). Improvements stemmed from a reduction in fertiliser and an increase in crop export, the former more likely a result of mitigation. Surplus reductions ranged from 15.6% - 67.0%, figures similar to those reported by Verbruggen et al., (2005) following efforts to improve nutrient management on Flemish farms. In line with Verbruggen et al., (2005) maximum improvements were observed on cattle and sheep farms. However, in contrast to the reduction in feed inputs observed in Belgium, improvements on MSA / EMEL farms predominately stemmed from a reduction in fertiliser. This reflects the fertiliser / manure focus of WAgriCo mitigation, and the absence of mitigation targeting feed management. Although not statistically significant, surplus improvements were more prevalent on farms adopting a higher level of mitigation. Similarly, larger improvements were witnessed on pilot farms adopting mitigation on a higher technological level (Langeveld et al., 2005), and lower surpluses reported where demonstration farm technology was implemented over more widely available technologies (Hilhorst et al., 2001).

The voluntary nature of WAgriCo mitigation and a lack of experimental control meant changes in nutrient use were smaller and less significant than those observed in designed farm / farmlet studies. Ledgard et al., (1999) for example investigated the impact of a 135kg N ha^{-1} (37.5%) reduction in fertiliser to grass on dairy farmlets; this is considerably larger than voluntary reductions on MSA dairy farms which averaged 17.9kg N ha^{-1} (12.1%) (fertiliser inputs to EMEL farms increased). Where implemented in a practical and voluntary context, mitigation is also subject to compensatory behaviour. While the reductions in fertiliser investigated in Ledgard et al., (1999) resulted in a 43kg N ha^{-1} (12.9%) reduction in the farmgate surplus, increased feed inputs meant surpluses increased on MSA farms. While catchment scale demonstration projects such as WAgriCo offer a more realistic insight into mitigation uptake and impact, in doing so they forego experimental control.

The black box nature of farmgate budgets meant reductions in fertiliser reflect responses to both fertiliser and manure mitigation. Similarly Verbruggen et al., (2005) and Vinther et al., (2009) suggest improved manure utilisation contributed to the 25 and 50% reductions in fertiliser observed on Flemish and Danish farms respectively. Where both manure and fertiliser mitigation was adopted, it was not possible to differentiate between the impact of individual methods with any certainty. Increasing feed inputs suggests mitigation failed to address feed N or have a whole farm impact; larger improvements might be achieved if mitigation was considered on a whole farm basis. Numerous studies highlight the importance of maximising feed efficiency and adjusting rations as a means to reducing farm surpluses (Berentsen and Tiessink, 2003; Borsting et al., 2003; Vinther et al., 2009). Farm scale budgets effectively capture compensatory behaviour meaning they are well suited to whole farm mitigation evaluations. Mitigation must however address all elements of nutrient use including feed to fully exploit this.

Subsequent analysis of annual nutrient use and farm characteristics highlighted that improvements between pre- and post-mitigation year surpluses provided no confirmation of mitigation sensitivity. Indeed improvements were not significant in MSA. Variability between 2005 and 2007 was found to exceed that between pre- and post-mitigation years and reductions in nutrient use occurred prior to the implementation of mitigation. In agreement with Ondersteijn et al., (2002), surpluses were found to vary not only between farms but also within farms due to farm layout (stocking density and cropping). Once changes in cropping and livestock had been accounted for, only the improvements observed on MSA cattle and sheep and EMEL cereal farms could be linked to mitigation; the improvements observed on MSA cereal and EMEL cattle and sheep and mixed farms were unlikely to have been mitigation driven. The calculation of hybrid ‘baseline' for MSA farms provided further evidence for these deductions, facilitating a direct comparison of crop nutrient management independent of cropping differences. While MSA cereal farm surpluses fell by more than 20% between ‘baseline’ and mitigation years, hybrid ‘baseline’ revealed 2008 cropping was likely to yield lower surpluses regardless of the mitigation implemented. The opposite was true on MSA dairy and mixed farms where 2008 cropping was likely to yield higher surpluses. Despite surpluses increasing between the ‘baseline’ and mitigation years, comparison on a like for like cropping and livestock basis revealed improvements on both farm types. Similar observations have been made by Langeveld et al., (2005) who attributed an increase in surpluses to changes in crop distribution following the dissemination of fertiliser recommendations to Dutch farmers.

When evaluated over relatively short timescales changes in cropping and livestock density will only be fully accounted for where budgets are calculated on a crop specific basis or presented relative to livestock density. However, in doing so the benefit of not needing crop specific management information is foregone. Where changes in feeding regime and crop export fraction fluctuate, Schroder et al., (2003) suggest the use of conversion efficiencies to quantify improvement more objectively. Efficiencies highlight how effectively nutrient are utilised and have been shown to increase despite increases in farm surpluses (Chapter 7; Swenson, 2003) and feed inputs (Nielsen and Kristensen, 2005). Focusing on efficiency ensures productivity is maintained whilst taking steps to minimise environmental impact.

Similar to field scale budgets, correspondence between high crop export and high yields in 2006 and 2008 (Chapter 5; Defra, 2009) exposed a connection between weather variability and farm nutrient balances. With crop export representing approximately 80% of N output from both catchments, and minimum catchments surpluses observed in these years, the significance of environmental variability on budget based mitigation evaluations is potentially large. This was also demonstrated by Verbruggen et al., (2005) who attributed low crop export and smaller improvements in surpluses on arable farms than beef / dairy to poor weather. Weather is also known to affect N fixation (Ledgard et al., 1999) and grass N content (Jurgen et al., 2006), the latter affecting feed imports and crops export. Nutrient management is therefore both responsive to and affected by changes in environmental conditions irrespective of mitigation.

Nutrient use is also affected by changing economic, political and social conditions: thus, farm budget assessments are confounded by yet more variability. In both catchments total fertiliser inputs decreased throughout the study period. While mitigation may explain some of this reduction, a corresponding increase in the price of fertiliser is likely to have had an influence on purchasing decisions. In addition, increased environmental awareness has meant increasingly stringent legislation affecting nutrient management in recent years, and where assessments overlap with regulatory change, evaluations of mitigation effectiveness are affected. WAgriCo farms however were included within the original designation of Nitrate Vulnerable Zones exposing them to minimal regulatory change during the study. However prior compliance with NVZ regulations may have reduced scope for further mitigation induced change.

Balancing the implications of short term weather variability against longer term socio-economic changes complicates the selection of optimum study length. While short term studies are not long enough to attribute change to mitigation with confidence (section 6.3.1; Halberg, 1999; Swenson, 2003), longer term projects are likely to encounter greater social and political change. Kyllingsbaek and Hansen (2007) demonstrated the extent to which policy can alter nutrient management over the longer term, reporting a 40% reduction in surpluses between 1980 and 2004 following the introduction of Danish National Action plans aimed at reducing N loss from agriculture; the long time series confirmed a downward trend in nutrient use despite annual variability in yield. Furthermore, responses to mitigation may not be instantaneous with implementation extending beyond one year and impact initially buffered by advance purchases of feed and fertiliser. Farms take time to reach new equilibria (Aarts et al., 2000). To account for the interactions between years, the calculation of moving averages is advocated where data allows (Kyllingsbaek and Hansen, 2007).

Table 6-11: Comparison of MSA and EMEL farmgate surpluses (kg N ha^{-1}) with results reported in the literature. Full details of contributing references can be found in table A-4 in the appendix.

Farm type	Observed Farmgate surplus (kg N ha^{-1})		Surpluses reported in literature (kg N ha^{-1})		
	MSA	EMEL	Average	Max	Min
Cattle and sheep	65	68	135	285	-18
Cereal	66	50	95	160	66
Dairy	232	202	218	318	117
Mixed	144	85	181	285	76

Insignificant differences in nutrient use and surpluses between years, coupled with changes in nutrient management attributed to other factors besides mitigation (namely changes in cropping and stocking rates) suggests farmgate budgets were insensitive to the mitigation methods adopted. However it is important to note that assessments were made over a four year period with only one year’s data post mitigation. The extent to which budgets captured differences in nutrient use between farm types and catchments suggests budgets have the potential to be sensitive to mitigation where mitigation drives a notable change in nutrient use. However over the timescales investigated mitigation driven change was modest, and annual variability stemming from environmental and economic change evident. Agreement with farm type average surpluses in the literature (Table 6-11) and confirmation of a positive relationship between livestock density and farmgate surpluses noted in the literature (e.g. Dalgaard et al., 2002; Defra, 2005) provides additional confidence in the sensitivity and robustness of the methodology to wider differences in nutrient use. Although results were largely insignificant, a tendency for farms to display lower surpluses post mitigation is encouraging. However perhaps more important was the level of interest with which results were received by farmers. Budgets may not have exposed significant responses to mitigation but confirmed their communicability. When present visually farmgate surpluses offered an understandable and interpretable way to characterise nutrient use at a scale of interest to farmers.

6.5.2 Farm scale mitigation scenarios

Mitigation scenarios confirmed that larger reductions in surpluses are achievable if more extensive mitigation was to be adopted more widely. This suggests the voluntary nature of WAgriCo mitigation, not the insensitivity of budgets to mitigation, limited positive responses to mitigation in MSA and EMEL. The integration of total farm RAN in 'baseline' fertiliser supply proved most effective at the farm scale, resulting in consistently larger reductions than WAgriCo mitigation. However, prior manure accounting and the sub-optimal application and utilisation of manure / excreta may limit the extent to which similar reductions are observed in reality. In contrast, fertiliser recommendations had a negative impact on all but dairy farms, highlighting discrepancies between PLANET fertiliser recommendations (Defra, 2000; Defra, 2006) and local application rates. Differences between the two fertiliser recommendation scenarios also confirmed that pre-mitigation manure accounting was highest on cereal farms, reducing scope for improved manure utilisation especially given low on farm manure production. Mitigation performance depended on the differences between actual and recommended fertiliser supply, total farm RAN and the extent of pre-mitigation manure accounting.

Responses to case study scenarios were comparable to those observed under wider mitigation scenarios, and resulted in consistently larger improvements than those witnessed following WAgriCo mitigation. However given that simulations were restricted to 'maximum impact' farms, results are likely to represent maximum benefit. Reductions in feed proved slightly less effective than reductions in stocking and arable area, and on the individual farm did not perform as well as catchment wide scenarios 4 and 5 owing to high RAN relative to feed imports. While simulated reductions in surpluses were similar to those observed by Kristensen et al., (1997) following a comparable reduction in feed N, Kristensen et al., (1997) observed compensatory behaviour; reductions in feed N led to increased fertiliser applications. Similar behaviour was considered too uncertain to simulate on MSA / EMEL farms meaning the benefit of feed N reduction may have been overestimated. Modelled scenarios would therefore benefit from more complete understanding of likely feedbacks and / or efforts to minimise compensatory behaviour. The impact of arable conversion and reductions in stocking density was similar to that of RAN accounting with surplus reductions similar to the level of changed induced. While these results are encouraging and are in agreement with the success of Nitrate Sensitive Area arable reversion (Lord et al., 1999), economic consequences are likely to be substantial. In contrast, the success of improved manure utilisation is coupled with the financial benefits of reduced fertiliser import.

6.5.3 Catchment scale results

Following the adoption of mitigation catchment scale surpluses derived from an enlarged farmgate methodology decreased by 24.4 and 5.9% in EMEL and MSA respectively. Improvements stemmed from a reduction in fertiliser use coupled with increased crop export post mitigation. Improvements were greater high in EMEL due to larger reductions in fertiliser and a reduction in feed; across the whole catchment feed inputs increased to MSA after mitigation. However observed improvements cannot be fully attributed to mitigation. In both catchments fertiliser inputs fell across all years, with maximum reductions occurring prior to the implementation of mitigation in 2005/6, and correspondence between maximum crop export and minimum surpluses highlighting the importance of output in governing catchment surpluses. Differences in cropping between years was found to explain the majority of variability in the MSA surplus, and while reduced feed inputs contributed to the improvements in EMEL, feed management was not directly targeted by mitigation.

Comparison of farm and catchment scale results highlighted the importance of farm size - the behaviour of a few large farms caused catchment and average farm results to deviate. But whilst accounting for farm area and farm type distribution facilitates inter-catchment comparisons, this must be conducted in the presence of farm type distribution data to ensure the implications of farm type on nutrient use, scope for improvement and responses to mitigation are considered.

6.5.3.1 Catchment scale mitigation scenarios

In agreement with farm scale results scenario 5 (Integration of manure RAN within 'baseline' fertiliser applications) proved most effective in reducing the catchment surplus, highlighting applicability across all farm types and widespread scope for better manure utilisation. Yielding a 43.5% reduction in the catchment surplus results were comparable to the 40% reduction in the national surplus observed in Denmark following the introduction of National Action Plans aimed at improved manure utilisation (Kyllingsbaek and Hansen, 2007). However, calculation of a catchment scale surplus concealed farm type and individual farm specific responses which must be considered to achieve maximum catchment scale impact; blanket mitigation should therefore be avoided. While catchment scale assessments are relevant to the catchment scale implementation and the waterbody focus of the WFD, aggregating farm results may conceal localised improvements / deterioration, which, dependant on equally localised environmental conditions, may have significant affects on water body status. The respective benefits of adopting farm or catchment scale evaluations depend on the relationship between surpluses and loss observed locally, and the geographical extent of the study. While there is little difference in the level of computation, catchment results are useful where inter-catchment comparisons are to be made.

6.6 Conclusions

Farmgate budgets exposed lowered surpluses on a large proportion of farms with inputs displaying initial downward trends however sensitivity to a range of other factors including production intensity, farm layout (cropping and stocking density), environmental conditions and socio-economic situations restricted the extent and certainty with which change / improvements could be attributed to mitigation. Farmgate surpluses must be interpreted relative to farm economics which drive or result in the changes observed in nutrient use besides mitigation. Sensitivity to differences in nutrient use between farm types and catchments coupled with large improvements under mitigation scenarios suggests more significant responses to mitigation would have been observed on MSA and EMEL farms had more drastic mitigation been implemented. The voluntary nature of WAgriCo mitigation not the insensitivity of farmgate budgets is likely to have limited more significant responses to mitigation.

Results demonstrated the appropriateness of farm scale assessments, capturing feed backs, accounting for complete crop rotations, (even when calculated on an annual basis) and respecting the applicability / compatibility of mitigation within crop rotations and farm systems. Yielding farm specific output, farmgate budgets were of relevance to farmers, who's interest and engagement is key to improvements in nutrient management.

Though it was not possible to evaluate individual mitigation methods due to the black box nature of farmgate budgets, farm and catchment scale mitigation scenarios highlighted opportunities for improved manure accounting, demonstrating its effectiveness both in terms of surplus reductions but also reduced fertiliser costs (through a reduction in fertiliser inputs). However mitigation scenarios also confirmed that mitigation performance depends on 'baseline' conditions i.e. scope for (further) improvement. Interpretation of changes in nutrient fluxes on a farm type basis highlighted the need for whole farm mitigation and in particular mitigation addressing feed management.

Farmgate surpluses were successfully applied to the catchment scale, facilitating inter-catchment comparison. Catchment scale surpluses must however be interpreted relative to farm type and size distribution, acknowledging the farm and farm type specific responses to mitigation, and changes in farm layout across the catchment. And although of relevance in the context of current legislation, catchment scale surpluses forego their appeal to farmers and usefulness in identifying high risk areas in which to target future mitigation. The latter is of particular poignancy given the need to ensure the WFD and associated PofMs are cost effective.

7 Using farm scale 'efficiency' to evaluate mitigation effectiveness

7.1 Introduction

7.1.1 Introduction to farm scale 'efficiency' and its use as evaluator of mitigation effectiveness

7.1.1.1 Efficiency and nutrient loss

Efficiency describes the ratio of useful output to total input for any system including that of agricultural systems (Halberg et al., 2005). The concept can be effectively applied to nutrient use, quantifying the proportion of imported nutrients in feed and fertiliser that are converted into useful products such as milk, meat and crops. However inefficient agricultural production systems mean not all inputs are converted into useful products leaving nutrients at risk of loss to the environment. As a result efficiency represents an alternative indicator of N use and loss to N budgets, and despite its non-direct relationship with environmental impact, is considered an effective agri-environmental indicator, for example the EALF indicator (Halberg, 1999; Halberg et al., 2005). Differences in efficiency have been shown to stem from differences in farm management meaning improvements in management can be expected to improve efficiency (de Koeijer et al., 2003). As a result efficiency has the potential to provide evaluations of mitigation effectiveness.

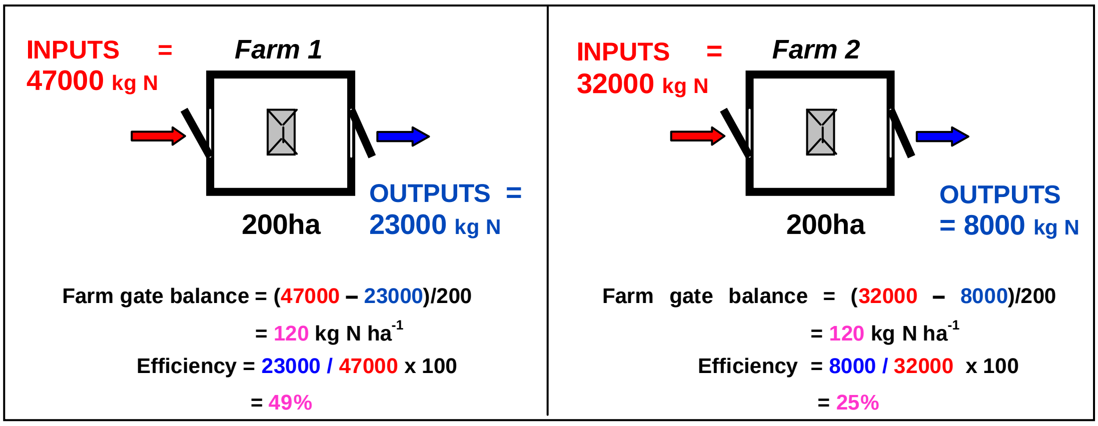


Figure 7-1: Differences between nutrient budget and nutrient efficiency methodologies. Farms 1 and 2 have the same surplus but different efficiencies.

7.1.1.2 Efficiency vs. surpluses

In contrast to N budgets which balance inputs and outputs, efficiency considers the amount of output produced for a given amount of input; it is therefore possible for two farms to have same surplus but different efficiencies (Figure 7-1). 'Efficiency' places greater emphasis on reducing losses through increased nutrient utilisation, not a reduction in inputs (Kohn et al., 1997; Rotz et al., 2005), and in doing so provides greater assurance of the economic sustainability of improved nutrient management (Koeijer et al., 2003). Making efficiency the focus of nutrient management avoids reductions in inputs which left unsupported may reduce productivity (Runge and Osterburg, 2007). Unlike surpluses, efficiency afford greater consideration to what is possible, recognising the inherent limitations of nutrient utilisation in crops and livestock (Van Bruchem et al., 1999; Lambert et al., 2005; Powell et al., 2009). However to some extent efficiencies compliment surpluses and the two are often calculated side by side. Corresponding improvements in efficiency and reductions in surpluses confirm production intensity has not been reduced, moreover the relationship between inputs and output has been altered and the utilisation of inputs improved (Nielsen and Kristensen, 2005).

7.1.1.3 Efficiency at different scales

Efficiency can be calculated at a range of scales, from sub-compartmental transfers e.g. feed N → livestock weight gain in cattle housing (Haas et al., 2002) to the whole farm (Treacy et al., 2008), and where feed is produced off farm, some argue evaluations be extended to whole production systems to account for inefficiencies in all stages of production (Borsting et al., 2003; Bleken et al., 2005). The purpose of the study often governs the scale of application, although in many cases whole farm efficiencies are accompanied by sub-compartmental analysis enabling improvements in individual transfers to be interpreted with respect to what is possible and to maximise productivity (Borsting et al., 2003; Lambert et al., 2005). Multi-scale analyses have also been conducted to evaluate change independent of the confounding influences of cropping and output price (Koeijer et al., 2003). However where the efficiency of subsystems are not connected, and where improvements in one elements reduces the efficiency of others, farm scale evaluations are favoured (Van Bruchem et al., 1999; Borsting et al., 2003; Bleken et al., 2005). Irrespective of the scale of calculation there is increasing awareness of the differing trophic levels accounted for in efficiency calculations, especially at the farm scale. Bleken et al., (2005) advocate the calculation of trophic level specific efficiency i.e. the efficiency of plant and animal production, however Jurgen et al., (2006) presented stall and grassland efficiencies favouring differentiation between specific farm components. Calculation of separate organic / mineral efficiencies also divides opinion. Webb et al., (2001) consider differentiation arbitrary, whilst Runge and Osterburg (2007) exploit subsequent opportunities to differentiate improvements in nutrient management from 'structural change' (changes in cropping and stocking which alter relative reliance on fertiliser and manure N).

7.1.1.4 Efficiency methods

Although all derived from the basic efficiency concept, a range of different efficiency indicator and methodologies exist. The simplest efficiency calculation (outputs / inputs) is most commonly utilised and has been effectively used to characterise nutrient use efficiency on different farm types by Domburg et al., (2000) and Bassanino et al., (2007). However Bleken (2009) encourages use of emission factors, a hybrid of surpluses and efficiency which afford greater consideration to production intensity by evaluating the ratio of (input – output) to output. In line with suggestions that the efficiency of individual transfers be evaluated to consider scope for improvement, Schroder et al., (2003) supports the use of conversion coefficients for each transfer. In doing so operational management and strategic farm structure is respected (for example reliance on off-farm produced feed / the export of crops) Similarly Reidy et al., (2005) and Lambert et al., (2005) consider what is biologically possible for a particular farm set up, calculating surpluses as a % of farm specific requirements, the latter derived from maximum achievable efficiencies for each transfer. However superimposed upon these differing equations is the modelled system which alike to budgets differs widely owing to data available and study purpose. It is therefore important that results be interpreted in the context of what each system represents.

7.1.1.5 Efficiency and mitigation

The role of efficiency in the context of mitigation is more varied than that of surpluses, fulfilling not only that of nutrient characterisation (Nielsen and Kristensen, 2005; Verbruggen et al., 2005; Treacy et al., 2008) and the tracking of changes in nutrient use (Nevens et al., 2006; Kyllingsbaek and Hansen, 2007), but supporting results orientated mitigation and providing communication and sustainability tools. Trialling the use of the MOTIFS graphical tool which integrates efficiency amongst many other indicators of economic and ecological sustainability, D'Haene and de Mey (2009) observed improved nutrient awareness, and attributed re-examination and subsequent improvement of nutrient management to an efficiency tool. As a means of mitigation, the achievement of target efficiencies / improvements in efficiency addresses the need for farm scale mitigation; action orientated mitigation is often implemented at the field scale, and the impact of these measures insufficient on high livestock farms (Runge and Osterburg, 2007). Adoption of results orientated measures are also likely to be less costly than equivalent action orientated approaches if implemented to the extent required for WFD compliance (Runge and Osterburg, 2007). Crucially goal orientated indicators afford farmers maximum freedom in selecting mitigation which is most (cost) effective under their specific circumstances yet achieving the desired goal (Van der Werf and Petit, 2002).

Central to the provision of results orientated mitigation is the availability of an indicator sensitive to management activities. By exploring the impact of management practices on farm and transfer efficiencies (Ledgard et al., 1999; Aarts et al., 2000), and evaluating the impact of environmental awareness campaigns on nutrient management (Swensson, 2003; Van Wepern, 2009), links between nutrient management and efficiency have been exposed. In an alternative approach, Kohn et al., (1997) conducted sensitivity analysis to confirm sensitivity to mitigation methods in particular. As a result Swiss legislation limits 'farm surpluses as a % of farm requirements' to a maximum of 10% above the calculated farm requirement acknowledging that better nutrient management will achieve efficiencies closer to theoretical maximums (Reidy et al., 2005). While the Netherlands adopted a surplus based results orientated approach, supplementary efficiencies have been calculated, providing assurance that production is not inadvertently affected (Ondersteijn et al., 2002).

In response to the benefits of results orientated mitigation identified by Runge and Osterbury (2007), and the expected sensitivity of efficiency to nutrient management, an efficiency based results orientated mitigation method was developed and implemented as part of the WAgriCo project (see section 3.5.1). Farmers in both Germany and the UK were rewarded for improvements in their nutrient efficiency on the premise that higher efficiency reflected better nutrient management.

7.1.1.6 Efficiency and mitigation evaluation

Despite considerable evidence that improved nutrient management achieves higher efficiency e.g. Nielsen and Kristensen (2005), and the understanding that implementing mitigation improves nutrient management, efficiency has rarely been used to evaluate the impact of specific mitigation. Previous results have tended to investigate changes in the production system but not compare mitigation effectiveness e.g. Ondersteijn et al., (2002); Swensson (2003); Kyllingsbaek and Hansen (2007). The sensitivity analysis conducted by Kohn et al., (1997) and the farmlet trials in New Zealand (Ledgard et al., 1999) identified where different mitigation options might have most impact and how higher levels of efficiency might be achieved, but neither attributed improvements to specific mitigation options. Given the communicability of the efficiency concept, links to wider economic sustainability and existing connections with mitigation, its usefulness of a means of evaluating mitigation effectiveness should be explored, and in doing so the respective merits of surplus and efficiency based indicators compared. Unlike surplus based investigations which yield a spatially distributed indicator, the scale applicability of efficiency is perhaps less flexible than for surpluses. Efficiency is more meaningful where it is attached to a discrete system and its application limited by the bounds of production systems. Despite the WFDs catchment focus, catchments do not operate as one system. As a result investigations into efficiency as a means of assessing mitigation effectiveness will be limited to the field / farm scale. However it is noted efficient farm systems remain the pre-requisite of efficient production catchment wide.

7.1.2 The current study

7.1.2.1 Hypothesis 1

‘Efficiency represents an effective method of mitigation evaluation’

7.1.2.2 Aims

Aim 1 - To develop an efficiency methodology suitable for the evaluation of mitigation effectiveness.

Aim 2 - To assess the sensitivity of the efficiency methodology to a range of mitigation methods.

Aim 3 - To compare efficiency and surplus based methods of mitigation evaluation.

7.1.2.3 Objectives

5. To develop a farm scale efficiency methodology suited to the evaluation of mitigation effectiveness. The approach will be based on a methodology developed in Germany as part of a results orientated mitigation option.
6. To explore the ability of efficiency terms to characterise farm type specific nutrient use.
7. To investigate the impact of mitigation on farm efficiency terms by:
    a. Comparing efficiencies before and after the implementation of mitigation on MSA and EMEL farms.
    b. Comparing responses to different levels of mitigation.
8. To compare 'efficiency' and 'surplus' based evaluations of mitigation effectiveness.

7.2 Method

Development of a UK mitigation focussed efficiency methodology was based on an existing methodology developed in Germany. The German methodology was devised to support the implementation of 'results orientated' mitigation and thus well suited to the evaluation of mitigation. However preliminary investigations indicated that not all nutrient fluxes were fully accounted for and that assumptions could be refined to make output more meaningful. In addition coefficients were derived from German farms which differ in terms of size, type and structure from those in the UK. The original methodology was subsequently modified to improve the value of output and to 'better suit' the approach to UK farms. The discussions which follow describe the process of evaluation and refinement of the original German methodology before presenting the development of a UK specific version.

7.2.1 The original German methodology

7.2.1.1 The Concept

The German ‘efficiency’ methodology builds on the simple efficiency calculation (output / input), applied at the farm scale on an annual basis. However it also attempts to calculate the efficiency of fertiliser (MinNefficiency) and manure N (OrgNefficiency) separately, documenting changes in nutrient use independent of structural change (Runge and Osterburg, 2007). Shifts in reliance between organic and mineral N are therefore considered, and the inherently lower efficiency of organic N acknowledged; as a result it aimed to reward only genuine improvements in nutrient management. In the context of mitigation sensitivity, independent calculation of organic and mineral N efficiency respects the scope and importance of improved manure utilisation especially on livestock farms (Runge and Osterburg, 2007) and the usefulness of more detailed assessments of mitigation effectiveness.

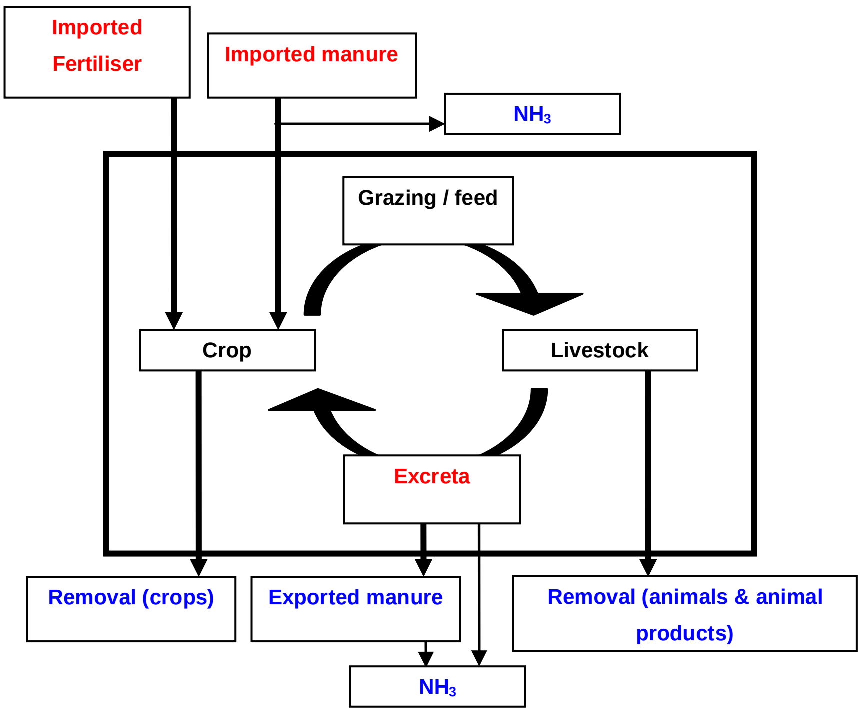


Figure 7-2: The farm system as modelled by the German version of the efficiency methodology (inputs are in red, outputs in blue) (Osterburg, 2008 pers. comm.)

Inputs and outputs in the German model are similar to those of farm scale budget methodologies, reflecting the main nutrient fluxes onto and off the farm (Figure 7-2). However unlike many existing methodologies excreta is also included, essentially replacing feed as an N input. Whilst this alters system boundaries and deviates from the more traditional farmgate budget, it explicitly accounts for on farm manure production and utilisation, and acknowledges the importance of excreta in linking herd and field activities (Van Bruchem et al., 1999; Borsting et al., 2003; Bleken et al., 2005). In contrast farmgate budgets adopt a 'black box' approach thereby offering no consideration to the internal cycling of nutrients. Given the comprehensive representation of excreta / manure, ammonia losses were also accounted for. An accountability factor of 0.6 was applied to manure retained on the farm respecting the relationship between manure N and ammonia loss. The methodology aimed to provide an indicator sensitive to management activities, and in doing so has the basis of an approach suited to mitigation evaluation. Full details of the calculations involved are presented in Tables 7-1 to 7-3.

7.2.1.2 The calculations

Table 7-1: Inputs and outputs of the German efficiency methodology

Input / Output	Fig reference	German Methodology terminology	Description	Calculation
Inputs	Excreta	Excreta gross	Total excretal returns (kg N)	= Σ (N content (kg head $^{-1}$ day $^{-1}$) x Number of stocking units x 365)
	Imported fertiliser	MinN	Imported fertiliser (kg N)	n/a
	Imported manure	OrgNimport[a]	Imported organic N (kg N)	= Σ Manure imports (kg N)
Outputs	Removal	Removal	3yr average of farm produce (kg N)	= Σ Exported crops (kg N) + Exported livestock (kg N) + exported animal products (kg N)
	Exported manure	OrgNexport[a]	Exported organic N (kg N)	= Σ Manure exports (kg N)
	NH3	n/a	OrgN*accountable* (fixed factor of 0.6) applied to organic N (see below)	

[a] net of NH^3 loss, determined via sample analysis

Table 7-2: Balance calculations in the German efficiency methodology

Balance	Description	Calculation
OrgN balance	Organic N retained on the farm + imported manures (total organic N on farm)	= OrgN*own* + OrgN*import*
		Where OrgN*own* = (Excreta Gross – OrgN*export*) x OrgN accountable
Gross Balance [a]	Farm balance gross of gaseous losses	= Excreta gross + minN + removed – OrgN*export* + OrgN*import*
Net Balance [a]	Farm balance net of gaseous losses	= OrgN + MinN + Removal

[a] Supplementary information – not required for calculation of efficiency

Table 7-3: Efficiency calculations in the German efficiency methodology

Efficiency	Conditions	Calculation
OrgN efficiency	OrgN > 0	= (Removal – MinN x MinN efficiency)/ OrgN
	OrgN < 0	= 0.4
MinN efficiency	OrgN > 0	= 1
	OrgN < 0	= (Removal / MinN)
Total N efficiency	n/a	= Removal / (OrgN + MinN)

7.2.1.3 Applicability to the UK

Preliminary calculations were undertaken to assess the suitability of the German efficiency methodology to MSA and EMEL farms. Results, although limited, highlighted very high mineral N efficiencies and low / negative organic N; total efficiencies were considered reasonable, if lower than expected on the cereal farm (Figure 7-3). Positive OrgN balances meant a default mineral efficiency of 100% was adopted on both farms, and given the inter-dependency of organic and mineral efficiency, high mineral efficiencies induced low organic N efficiency.

To ensure comprehensive characterisation of N use and loss, N flows were visualised in conceptual diagrams (Figure 7-4 - Figure 7-7). Flow diagrams highlighted missing N where feed inputs are high. Excreta does not fully account for N consumed in feed, indeed it represents only that which is not utilised.

A summary of the limitations associated with the German methodology with respect to the evaluation of mitigation effectiveness on UK farms are presented in Table 7-4, accompanied by possible improvements. Where refinement involved numerous iterations and re-evaluation (efficiency calculation, accountability, and system boundaries), further details are provided in section 7.2.2.

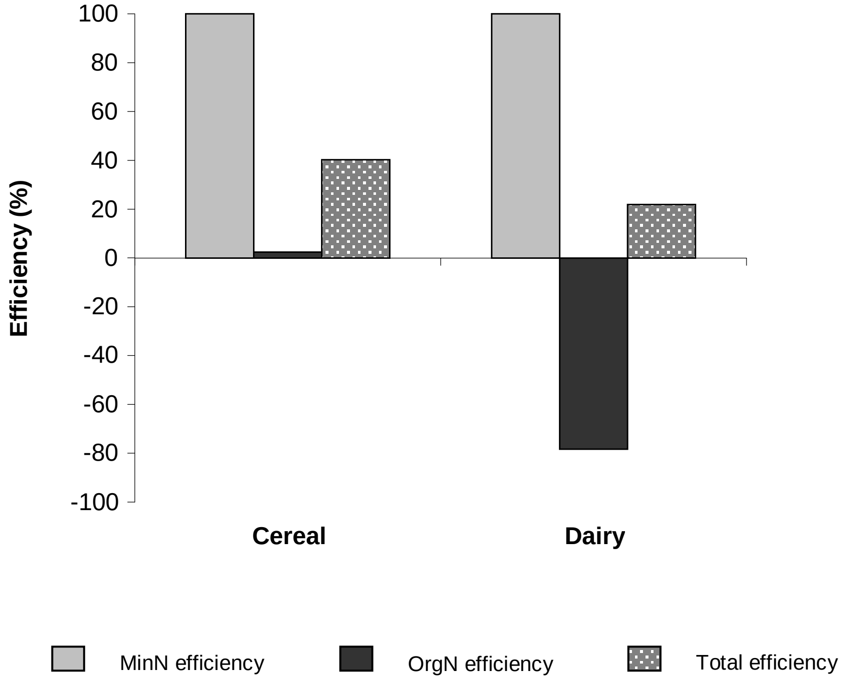


Figure 7-3: German efficiency methodology applied to selected MSA and EMEL farms

Table 7-4: Assessment of the suitability and applicability of the German efficiency methodology to UK farm systems.

Methodology feature	Limitation / area of concern	Action
System boundaries / inputs and outputs	Does excreta account for all N consumed in feed?	Include feed N as an input, integrating in mineral efficiency and organic N efficiency calculations. Care must be taken to avoid double counting with excreta especially in total efficiency.
	Since excreta is included should grazing / consumption of feed crops by livestock be considered more explicitly?	Integrate feed efficiency into mineral efficiency where grain retained on farm / grass grazed.
Min N efficiency	Where OrgN > 0, Min N efficiency was, by default, 100%.	100% mineral efficiency too high. Replace default value / calculate both OrgN and MinN efficiency.
	High min N efficiency meant Org N efficiencies were low or negative (e.g. Figure 7-3)	
Org N efficiency	Where OrgN < 0, Org N efficiency was, by default, 40%.	Adjust default value
Accountability (NH3 loss)	Is an accountability factor of 0.6 applicable to UK farm systems?	Develop comparable UK value
	Are all NH_3 losses accounted for?	NH3 loss from imported and exported manure determined via analysis – similar values not available in the UK.
Timescales	Removal calculated on a 3 year average basis which is inconsistent with budget methodologies in chapters 5 and 6.	Include year specific removal. To account for annual variability a 3 year pre-mitigation average will be calculated.

7.2.2 Development of a UK efficiency methodology

7.2.2.1 Inputs, outputs and system boundaries

Feed was added to the system inputs to account for missing N. However to ensure N was not double counted through inclusion of both feed and excreta, total efficiency was calculated using feed N and not excreta N; excreta is an internal transfer which does not need be considered when calculating total efficiency (it is however considered when calculating manure efficiency- see below) (Figure 7-4). While the resulting representation is now similar to a farmgate budget, accurate accounting of N makes this a necessity. With respect to mineral (fertiliser) N efficiency, feed was included as a sub-compartment, represented by a total input and an efficiency value alike to fertiliser (Figure 7-5 and Figure 7-6) (see section 7.2.2.2 for discussion of fertiliser efficiency values). When calculating manure (organic N) efficiency, fertiliser and feed removal are both subtracted from the total removal. However to retain the explicit accounting of manure N use, excreta remained an input in manure efficiency calculations. As the fraction of feed *not* converted into useful product, excreta N is available for utilisation and conversion into useful products without double counting with any of the removal attributed to feed (Figure 7-7). Its own utilisation can therefore be an element of manure efficiency.

Total efficiency

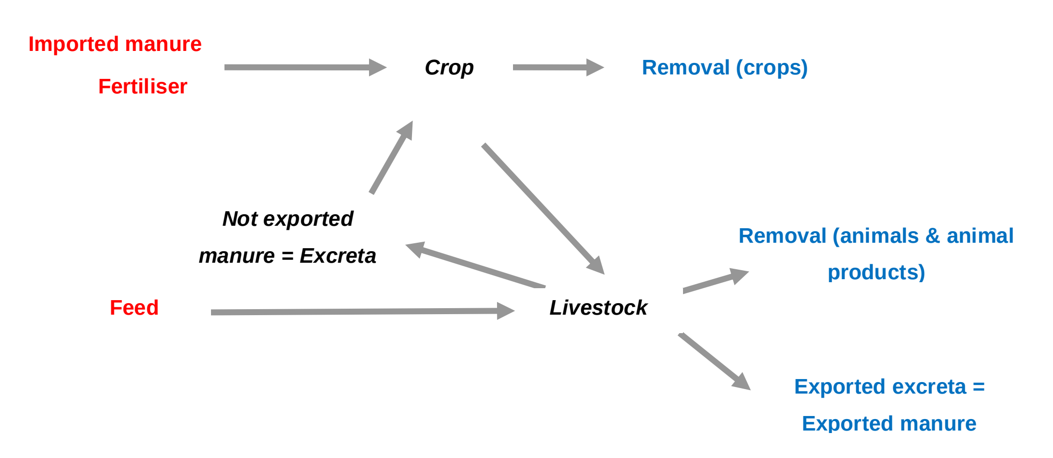


Figure 7-4: Flows and components of total efficiency. Inputs in red, outputs in blue, internal transfers in black.

7.2.2.2 Fertiliser (minN) efficiency

The default mineral efficiency of 100% (when OrgN >0) in the original methodology was considered too high given the fertiliser use efficiencies observed in the literature (e.g. Sylvester-Bradley and Kindred, 2009). It was hoped that the mineral efficiency could be calculated in a similar way to that of manure efficiency; the fraction of removal attributed to one N source (based on its magnitude and efficiency) subtracted from the total removal thereby facilitating calculation of the remaining N source's efficiency. However the mutual dependency of the two result in unsolvable circular functions and a compromise was sought. It was decided that one value would be derived from literature and the other calculated from farm data as described above.

Although affected by application rate, crop type, soil type, climate and cultivation (Webb et al., 1998; Sieling et al., 1998a and b; Nissen and Wander, 2003; Sieling and Kage, 2006; Giambalvo et al., 2009), fertiliser efficiencies avoid issues of manure N availability which are superimposed upon factors influencing mineral N utilisation (Sieling et al., 1998a and b; Webb et al., 2001; Powell et al., 2009; Sorensen and Thomsen, 2009). In addition there remains greater scope for improved manure utilisation, necessitating farm specific calculation. As a result the use of standard fertiliser efficiencies over standard manure efficiencies was favoured.

Crop nutrient use efficiencies (grain N / fertiliser N) were chosen over nutrient uptake efficiencies (grain + straw N / fertiliser N) on the grounds that grain N is the main useful crop product, and to simplify calculations where grain and straw from the same crop have different fates. For consistency straw was also removed as a farm output. To account for differences in cropping, crop specific values were obtained, and the relative contributions of each crop calculated on an area weighted basis. Although many values are available in the literature, results from Sylvester-Bradley and Kindred (2009) were chosen owing to the broad range of crops considered and the UK origin of results (see table A-5 in the appendix for full details).

7.2.2.3 Feed efficiency

In order to account for removal attributed to feed (a new input in the modelled system), feed efficiency values were required (Figure 7-5). Following difficulties in calculating more than one efficiency component from farm data (discussed above with reference to fertiliser vs. manure) and with WAgriCo mitigation not directly addressing livestock feed management, it was decided that alike to fertiliser, feed efficiencies would be sourced from the literature.

Feed efficiency

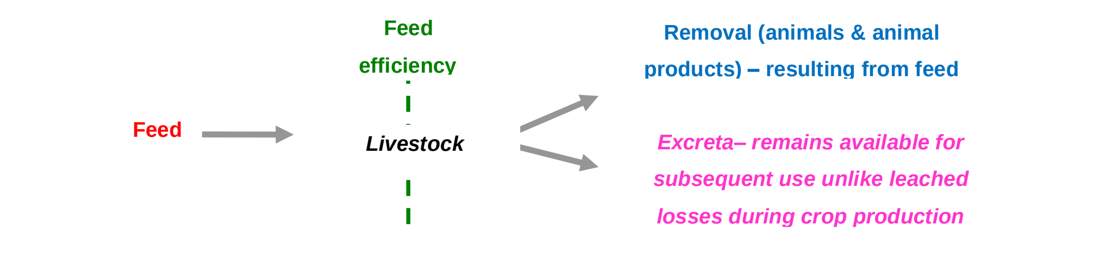


Figure 7-5: Flows and components of feed efficiency. Inputs in red, outputs in blue, internal transfers in black, conversion efficiencies in green.

Fertiliser efficiency

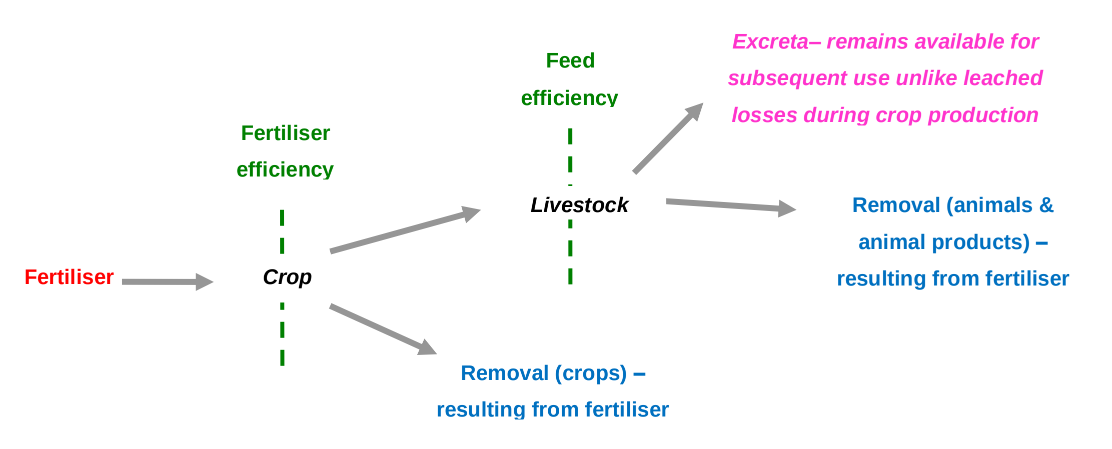


Figure 7-6: Flows and components of fertiliser efficiency. Inputs in red, outputs in blue, internal transfers in black, conversion efficiencies in green.

It was also decided feed efficiency would be integrated into calculations of fertiliser efficiency to account more fully for the transfer of N where crops were consumed on farm or grass was grazed and thus where livestock represents the final product

(Figure 7-6). Fertiliser efficiency was therefore divided into two sections, one reflecting the efficiency of fertiliser applied to exported crops, based solely on the fertiliser efficiency of crops, and the other calculating the efficiency of fertiliser applied to crops fed to livestock or grazed grass combining both the crop fertiliser efficiency and the efficiency of the animal consuming it. The contribution of each section to the overall fertiliser efficiency was determined by the relative proportion of fertiliser which passes through each pathway.

Crop specific fertiliser efficiencies were area-weighted in both components, with exported and retained crop areas deduced by dividing exported / retained quantity by the average yield. Livestock feed efficiency was weighted according to the relative number of each livestock type. Exported / retained fertiliser was apportioned according to the proportion of total grain exported / retained, assuming yield and fertiliser applications were uniform across all fields in that crop.

7.2.2.4 Manure (OrgN) efficiency

Following the introduction of feed, removal attributed to feed was added to the manure efficiency calculation (Figure 7-7). However as explained in section 7.2.2.1, excreta remained an input in this equation. Adjustments were also made to default values where total manure was N < 0. Rather than returning a value of 40%, the absence of manure justifies there being no manure efficiency value.

Manure efficiency

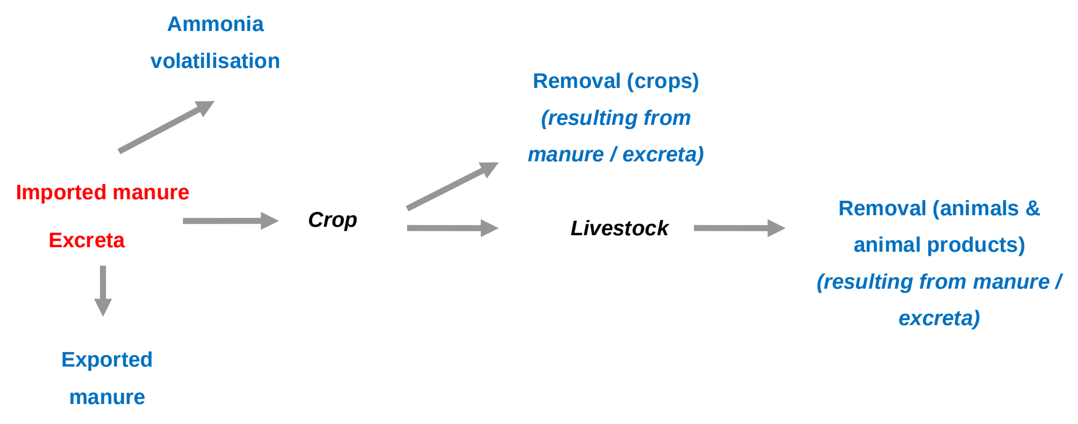


Figure 7-7: Flows and components of manure efficiency. Inputs in red, outputs in blue, internal transfers in black.

7.2.2.5 Accountability

Agriculture represents a significant source of ammonia (NH_3), accounting for approximately 90% of the total ammonia emissions in the UK (Misselbrook et al., 2000). With annual emissions estimated at 284.9 kt in 2004 (Misselbrook et al., 2006), retention of an accountability factor was favoured, however given differences in farm structure, management and climate between Germany and the UK, the availability of UK figures were explored. Initially it was decided that losses during manure spreading would be accounted for using MANNER (Chambers et al., 1999), a decision support system (DSS) which calculates ammonia loss according to manure type and application method / delay, which are known to affect ammonia volatilisation (Chambers et al., 1999). Losses from excreta, and the resultant excreta N inputs were sourced from N output standards collated during DEFRA project WT0715NVZ (Smith and Cottrill, 2007). Ammonia losses from both housing and during storage were quantified on a livestock specific basis. The modified approach meant ammonia losses and excreta N values were derived from UK studies and provided more detailed estimations of ammonia loss. However in making these changes the simplistic accountability factor approach was foregone and the need to input area weighted manure application rates and associated application details into additional software introduced.

In an effort to return to the original accountability factor format, the possibility of developing UK accountability factors was investigated. Using data from Smith and Cottrill (2007) the % of excreted N lost via housing and housing + storage was calculated on a livestock specific basis. In doing so it was possible to calculate the fraction remaining at each stage, termed ex-house and ex-store accountability factors. To account for spreading losses, a number of methods were trialled, two based on additional data found in Misselbrook et al., (2006) and one utilising MANNER (Chambers et al., 1999); details of the three approaches are provided in Table 7-5. The calculated spreading losses were subtracted from ex-store values resulting in ex-spread excreta N values. Comparing the original excreta value to the ex-spread value, ex-spread accountability factors were derived. To reduce the number of values, results were averaged by livestock type and are shown in Table 7-6 for the three ex-spread methods.

Table 7-5: Methods of accounting for ammonia losses during manure spreading. Methods AM0127_m1 and AM0127_m2 derived from Inventory of ammonia emissions from UK agriculture in 2004 (Misselbrook et al., 2004). MANNER method based on ammonia losses integrated within MANNER decision support tool for manure N accounting (Chambers et al., 1999).

Name / reference	Description	Assumptions / Limitation
AM0127_m1	Utilises a ratio between storage and spreading losses to calculate spreading loss. With storage losses = 13% of total NH3 loss and spreading loss = 29% of total nh3 loss (Misselbrook et al., 2006), spreading losses can be calculated by dividing the storage loss by 13 and multiplying by 29.	The relationship is not livestock specific and assumes high storage loss = high spreading loss inducing some negative ex-spread values. Justifiable only when using storage loss averaged across all livestock types.
AM0127_m2	Spreading loss derived from emission factors (EF). Spreading losses equivalent to 37 and 81% of total ammoniacal nitrogen (TAN) for slurry and FYM. Prior conversion of manure total nitrogen (TN) to TAN required; 50% and 10% of slurry and manure TN as TAN respectively. For poultry EF = 63% UAN which represents 40% of the TN. EF sourced from Misselbrook et al,. (2006), TAN – TN relationship from MANNER (Chambers et al., 1999) and RB209 (Defra, 2000).	More specific than AM0127_m1 but assumes the relationship between TAN and TN is linear; as ammonia is lost, the relative proportion of TN available for decreases.
MANNER	Typical manure application scenario inputted into MANNER for each manure type (200kg TN ha^{-1} applied on 1st November not incorporated on clay loam soil over chalk. Default rain and drainage applied). Output includes volatilisation loss.	Assumes a linear relationship between the TN application rate and NH_3 loss.

[a] The effect of manure incorporation delay on spreading losses and accountability is explored further in section.

[b] Soil type, rainfall, drainage and application date have no effect on volatilisation losses according to MANNER (Chambers et al., 1999)

Table 7-6: UK accountability factors for different manure types, for different accounting methods (see Table 7-5) and for different stages of production. Ex-house quantifies proportion of N remaining after losses livestock housing. Ex-store refers to losses during housing and from manure storage. Ex-spread refers to losses from housing, storage and spreading to land.

Stock type	Ex-House	Ex-Store	Ex-Spread (AM0127_m1)	Ex-Spread (AM0127_m2)	Ex-Spread (MANNER)
Cattle	0.90	0.86	0.77	0.74	0.73
Sheep	0.91	0.90	0.88	0.83	0.84
Pigs	0.90	0.71	0.26	0.61	0.63
Poultry	0.74	0.59	-0.01	0.41	0.43
Mean	0.88	0.76	0.50	0.65	0.66
German Accountability					*0.60*

With the exception of AM0127_m1 pig and poultry, ex-spread values were similar across the three methods. However, given the overly generic relationship utilised in AM0127_m1 and the questionable relationship between TAN and TN with increasing time, the MANNER method was preferred. Furthermore, the presence of

MANNER in the public domain provides greater assurance of reliable results. In the wider context of good agricultural management, inclusion of an ex-spread factor also provided an opportunity to investigate the impact of improved manure incorporation on efficiency, and ensure improvements in manure handling translate to reduced inputs; failure to account for such improvements could inadvertently increase leached losses (Rotz et al., 2005). Although not the main focus of WAgriCo which addressed nitrate loss, reducing ammonia loss would further improve manure utilisation and improve the overall sustainability of farm systems. Accounting for reductions in ammonia loss would ensure leached losses did not increase through failure to While various studies highlight the abatement efficiency of rapid incorporation / deep injection and would provide figures for integration in AM0127_m1 and m2 (e.g. Chambers et al., 2000), MANNER offers a more comprehensive consideration of manure incorporation, further supporting its selection. Ex-spread losses were recalculated under a range of incorporation delays, the results of which are shown in Table 7-7.

Average UK accountability factors (across all livestock types) were slightly higher than those originating from the Germany methodology corresponding with national ammonia inventories which report larger losses from animal husbandry in Germany than the UK on a per ha basis (Berg et al., 2003; Misselbrook et al., 2006). Accounted ammonia losses are of a similar magnitude to those used to calculate farm nutrient requirements on Swiss farms which ranged from 15-40% dependent on livestock category and housing system (Reidy et al., 2005). Webb et al., (2001) reported a similar rank of livestock specific losses based on UK data, however values represent storage and housing losses only. With respect to ammonia abatement, reductions in N loss following rapid incorporation / injection are similar to those presented by Sonneveld and Bos (2009) under low emission techniques in the Netherlands.

Table 7-7: Effect of incorporation delay and manure type on accountability factor

Stock type	**Not inc.**	**6-10 days**	**< 24hrs**	**< 6hrs**	**Deep injection**
Cattle	0.73	0.74	0.77	0.82	0.86
Sheep	0.84	0.85	0.88	0.89	0.90
Pigs	0.63	0.64	0.66	0.69	0.71
Poultry	0.43	0.47	0.51	0.52	0.52
Mean	0.66	0.67	0.7	0.73	0.76
German Accountability	*0.60*				

The efficiency spreadsheet returned to its previous format, summing farm excreta on a gross (before ammonia loss) basis before applying the appropriate accountability to calculate the fraction of this which remained on farm (excreta gross – exported manure). Owing to differences in the availability of manure sampling and N analysis between German and UK (parts of the project), imported and exported manure could not be offered on a net (after ammonia loss) basis via analysis as per the original German approach. While ammonia losses from imported and exported manure could have been accounted for using spreading loss only factor and storage and housing loss only factors respectively, this assumes prior consideration of storage and housing losses in farm input / output data. However farm data was far cruder than this and unlikely to have been adjusted to reflect prior housing and storage losses. It was therefore decided that an ex-spread factor would be applied to manure used on farm (excreta – exported manure + imported manure), but not to exported manure on the grounds that ammonia losses will be more affected by manure management on the receiving farm than on the exporting farm, and that the reduced N content should be considered at the time of utilisation not export.

7.2.3 The final efficiency methodology

Flows and calculations associated with the modified efficiency methodology are presented in Figure 7-8 and section 7.2.3.1 respectively. Further preliminary calculations were conducted to confirm the balance functioned correctly and that results are more reasonable than those obtained using the German methodology. As Figure 7-9 shows, mineral (fertiliser) efficiencies were now considerably smaller than those derived from the German approach, better reflecting the inefficiency with which mineral N is utilised, especially through the feed – livestock transfer. Organic (manure) efficiencies were notably higher, and negative results removed. In the case of the cereal farm, manure efficiency exceeded that of fertiliser, reflecting modifications made to ammonia accounting. The modified methodology accounts for all (housing, storage and spreading) ammonia loss from imported manure which will on some farms represent a significant reduction in net manure relative to the German approach. Given the substitution of excreta for feed, total efficiency was higher under the UK methodology; while the exclusion of feed in the German methodology was thought to leave some N unaccounted for (excreta N only accounts for feed N which is not utilised in meat or livestock products), feed inputs were generally lower than excreta inputs. Overall, results were considered more meaningful and thus provided additional assurance that the modified methodology was more appropriate and robust than the original one.

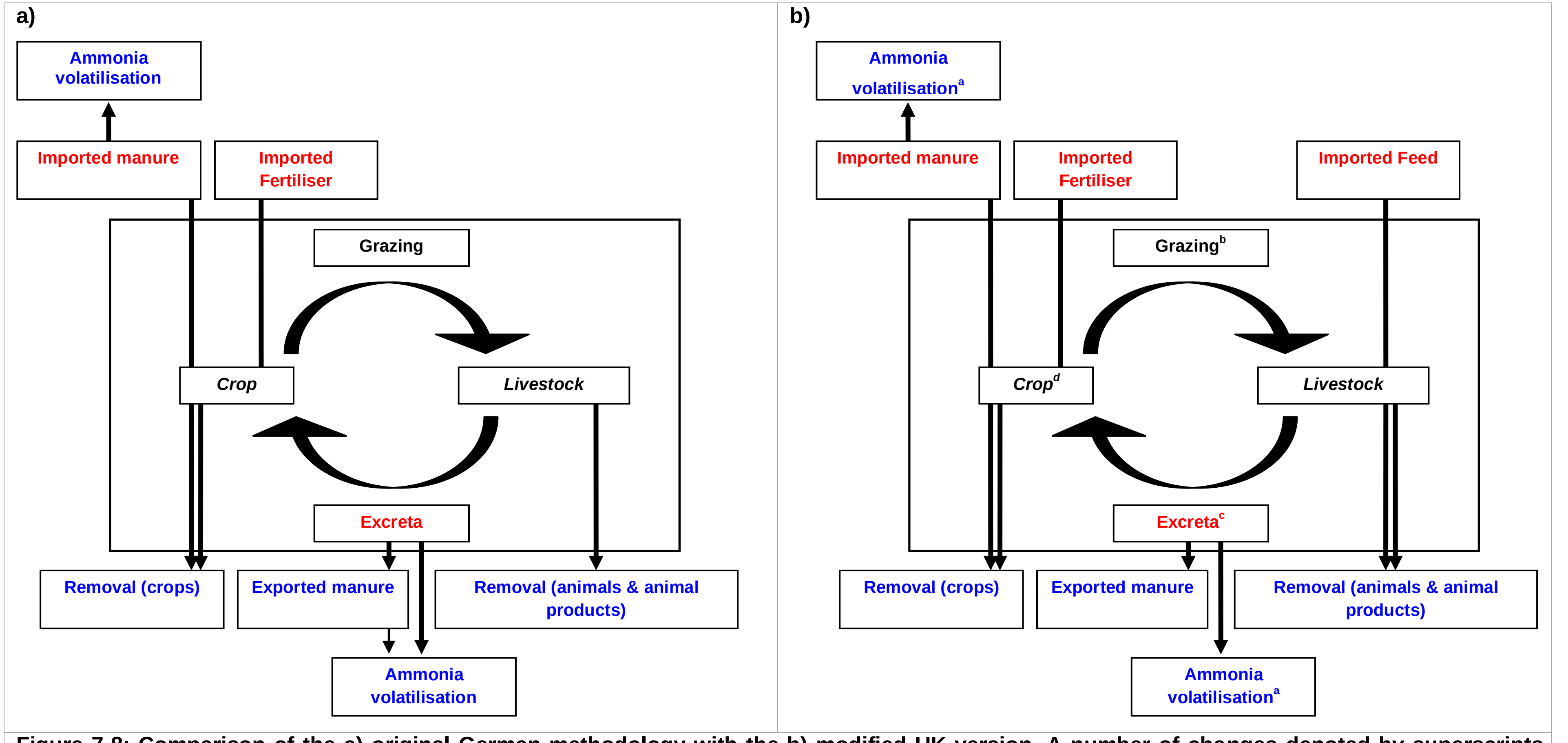
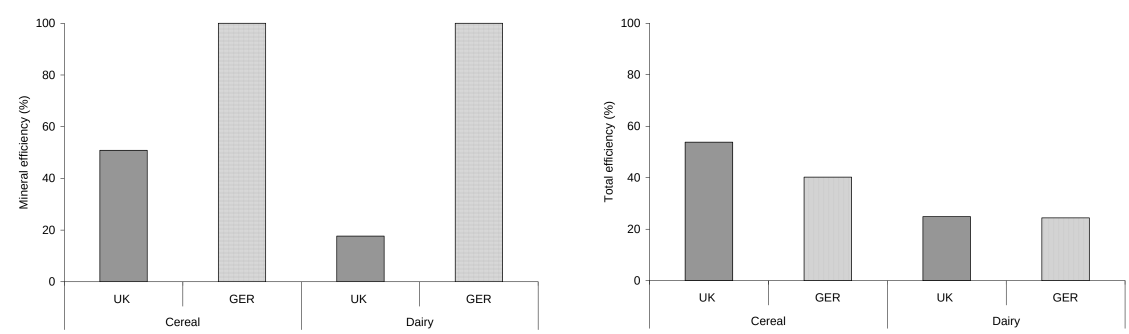


Figure 7-8: Comparison of the a) original German methodology with the b) modified UK version. A number of changes denoted by superscripts affect specific calculations only and thus not fully apparent in these diagrams: [a] Ammonia volatilisation calculated using UK specific accountability factors; [b] Feed → livestock / livestock products transfer accounted for in feed and fertiliser efficiency calculations; [c] Excreta represents an input to manure efficiency and not total efficiency; [d] Fertiliser efficiency is crop type specific.

a) b)



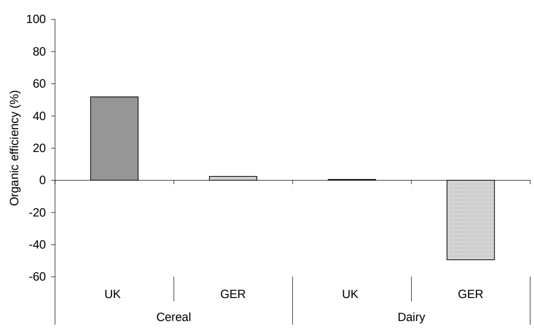


Figure 7-9: Preliminary results comparing the modified UK efficiency methodology to the original German methodology. a) Mineral (fertiliser) efficiency (%), b) total efficiency (%), c) Organic (manure) efficiency (%).

7.2.3.1 Calculations

Table 7-8: Inputs and outputs of the modified efficiency methodology

Inputs / Output	**Name**	**Calculation**
Input	Imported fertiliser (kg N)[a]	= Σ (Fertiliser applied to each crop (kg N))
	Imported manure (kg N)	= Σ (Manure import (t) x N content (kg t fw^{-1}))
	Feed (kg N)	= Imported feed N (kg N)
	Excreta (kg N)	= Σ (Gross excreta rate (kg N head $^{-1}$ yr $^{-1}$) x Number of stocking units x occupancy (%) / 100))
Output	Removal (crops) (kg N)	= Σ (Crop export (t) x N content (kg t fw^{-1}))
	Removal animals and animal products (kg N)	= Σ ((No. stock units x weight (kg) /1000 x N content (kg t fw^{-1})) + Σ (Quantity of animal product x weight (kg) / 1000 x N content (kg t^{-1})
	Exported manure (kg N)	= Σ (Manure export (t) x N content (kg t fw^{-1}))
	Ammonia	Relevant UK ex-spread accountability factor applied – see Manure N balance below

[a] Termed Fertiliser N in subsequent equations

Table 7-9: Balance associated with the modified efficiency methodology

Balance	Calculation
Manure N	= (Imported manure + excreta – exported manure) x ex-spread accountability factor)

Table 7-10: Efficiency calculations associated with the modified efficiency methodology

Efficiency	Calculation
Feed N efficiency	= Σ (Total livestock type LU / Total LU x Livestock feed N efficiency)
Fertiliser N efficiency	See below
Manure N efficiency	= (Removal – ((Fertiliser N x Fertiliser N efficiency) + (Feed N x Feed N efficiency)) / Manure N x 100
Total N efficiency	= Removal / (Fertiliser N + Feed N + Imported manure – Exported manure) x 100

<u>Fertiliser N efficiency</u>

$$= \text{‘Exported crop’ component} + \text{‘Retained crop’ component}$$

$$= \left(\frac{\text{Ave. crop fertiliser uptake efficiency for exported crops (\%)} \times \text{Fertiliser applied to exported crops (kg N)}}{\text{Total fertiliser (kg N)}} \right) + \left(\frac{\text{Ave. crop fertiliser uptake efficiency for retained crops and grass (\%)} \times \text{Average animal N efficiency (\%)} \times \text{Fertiliser applied to retained crops (kg N)}}{\text{Total fertiliser (kg N)} \times 100} \right)$$

Where…

Average exported/retained crop fertiliser efficiency (%) = Σ (exported/retained crop area (ha) / total crop area (ha)) x crop fertiliser uptake efficiency (%)

Exported/retained crop area (ha) = Exported/retained produce (t) / average yield (t/ha)

Average animal N efficiency (%) = Σ (Livestock LU / Total LU) x Animal feed efficiency (%)

Fertiliser applied (kg N) = Σ (total fertiliser applied to each crop (kg N) x produce (t) / total produce(t))

7.2.3.2 Assumptions

To minimise data requirements and structural uncertainty, a number of assumptions have been accepted.

- Fertiliser is assumed to be applied evenly to each crop.
- N content and yield is constant across the farm; exported and retained crops are therefore indistinguishable.
- Crops not receiving fertiliser have no affect on fertiliser utilisation and are therefore excluded from fertiliser efficiency calculations.
- Bedding and animal imports are small and show little annual variability. Impact on mitigation evaluations is likely to be negligible and subsequently remain excluded from the modelled system.
- In line with the farmgate methodology investigated in chapter 5, atmospheric deposition is excluded from the modelled system. This is on the grounds that the farmer has no direct control over atmospheric inputs, and being calculated on an area weighted basis would have no affect on mitigation evaluations. N fixation has not been included as an input despite inclusion in farm gate balances. In the case of clover rich grass, excreta will indirectly account for inputs, whilst the prevalence of leguminous crops was so low that addition calculations would generally be redundant.
- Calculations and data are year specific in line with farm gate budgets. Excreta is assumed to be produced and used in the same year, and only fertiliser applied to crops in that year is included (in contrast to farmgate budgets). Where activities in one year implicate on the next, or production extends beyond a single year (in some livestock production systems) this will be buffered through calculation of a 3 year average prior to mitigation.
- Manure samples and N analysis were not available for all farms meaning standard N contents for manure are used and ammonia losses accounted for using accountability factors.
- Gaseous losses (NH_3, N_2, and N_2O) from fertiliser are not accounted for on the grounds that they are small and are likely to have little effect on mitigation evaluations.
- All excreta is included, including that deposited during grazing. To reduce complexity no consideration is given to the uneven distribution and therefore suboptimal utilisation of excreta deposited to grass during grazing.
- Crop fertiliser uptake efficiencies are affected by the environmental conditions under which they are grown. However it has not been possible to source

comprehensive datasets under a range of different soil types, geology and climate conditions. A generic crop value has therefore been adopted. Impact on mitigation evaluation will however be reduced where conditions are site specific and vary very little between years.

- Adoption of a literature based fertiliser efficiency value assumes fertiliser is more effectively utilised than manure; this is highly likely given the cost of fertiliser. However where this is not the case, the inherent lack of sensitivity of fixed values to improvements in nutrient management means responses to mitigation will not be fully reflected in output. Given the interdependency of fertiliser and manure efficiency calculations a better solution could not be identified.
- Where excreta is deposited on grass fields (via manure spreading or during grazing), it is assumed this grass is cut and fed to livestock in the following year. If consumed in the same year nutrients would be double counted.

7.2.4 Data interpretation and analysis

The 'efficiency methodology' was developed as a standalone methodology organised within an Excel spreadsheet. Efficiency results were calculated on an annual basis for all farms for which data was available and mitigation adopted in the final year of the study. This represented a maximum of 34 farms in any one year (see Table 3-3). Further interpretation and analysis followed a very similar pattern to that of farm gate surpluses, calculating before and after results and testing for significant differences between inputs, outputs and efficiencies between farm types, catchment and years. The reader is therefore directed to section 6.2.5 for further details.

7.3 Results

7.3.1 Effect of mitigation on farm efficiency

Following the implementation of mitigation, total efficiency increased on 26 out of 34 farms (76.5%) with improvements across the 26 'improved farms' averaging 26.9%. Of the 32 farms using manure, 27 farms (84.4%) achieved higher manure efficiency post mitigation with improvements averaging 31.6%. However, significant differences in efficiency (total efficiency only) before and after were limited to EMEL farms (Figure

7-10 and Figure 7-11). Individual inputs and outputs, as well as manure efficiency displayed no significant difference before and after mitigation in either catchment (Table 6-5, Table 7-11). Substantial changes in nutrient use have still been noted in the discussions below because of the short time scale over which assessments were made. Fertiliser and feed efficiency are derived from the literature with differences before and after mitigation reflecting changes in cropping and livestock and not nutrient management. As a result changes in fertiliser and feed efficiency are not discussed in detail nor is supporting data shown.

Total and manure efficiency were not significantly different between catchments, and nor were responses to mitigation catchment specific (no significant interactions involving catchment). However inputs, outputs and efficiencies were significantly different between farm types, and for feed, crops, animal products and manure export, significantly different between catchments (Table 6-5, Table 7-11). Further discussion is therefore presented on a farm type- catchment specific basis. Significant relationships between stocking rate / cereal area and fertiliser efficiency were also observed (Figure 7-12 and Figure 7-13), supporting the differences in efficiency between farm types.

(Inputs and outputs included in total efficiency are also accounted for in farmgate budgets. To avoid duplication of results, reference is made to figures / tables in chapter 6 where changes in inputs and outputs have been shown.)

MSA Cattle and sheep farms

Total efficiency on cattle and sheep farms in MSA was low averaging 32.8%, a result of low (livestock) output relative to inputs (Figure 6-3 and Figure 7-10). Fertiliser efficiency was also low averaging 14.8%. This stemmed from high crop retention and low feed – meat conversion by beef cattle. Manure efficiency was of a similar magnitude to total efficiency, averaging 30.7% (Figure 7-11).

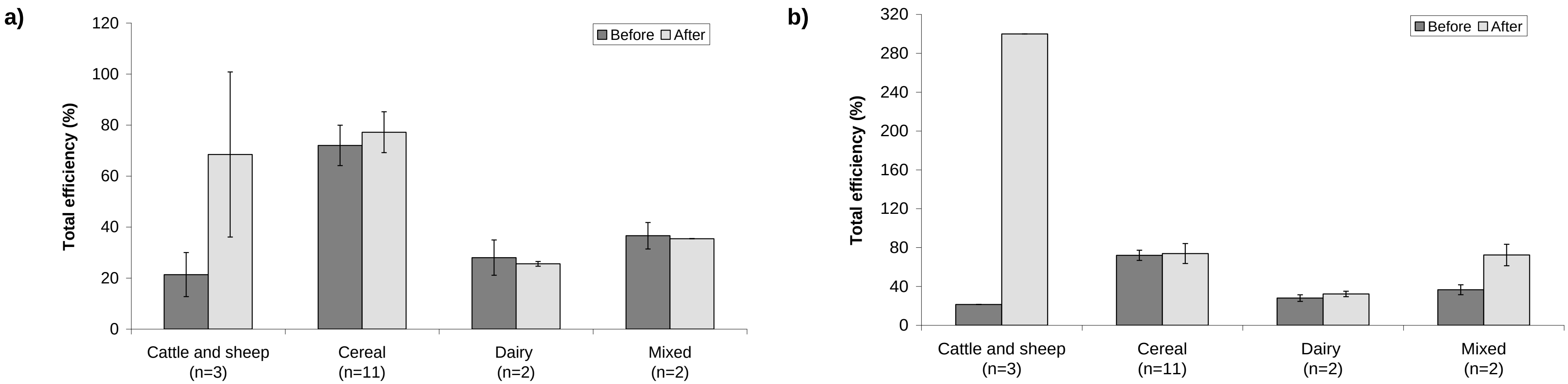


Figure 7-10: Total efficiency (%) before and after the implementation of mitigation in a) MSA and b) EMEL. Mean and SE shown with sample size in parentheses. Differences between years significant to p<0.05 in EMEL only. Differences between farm types significant to p<0.001 in both catchments. Differences between catchments not significant. Interactions between farm type, year and catchment not significant.

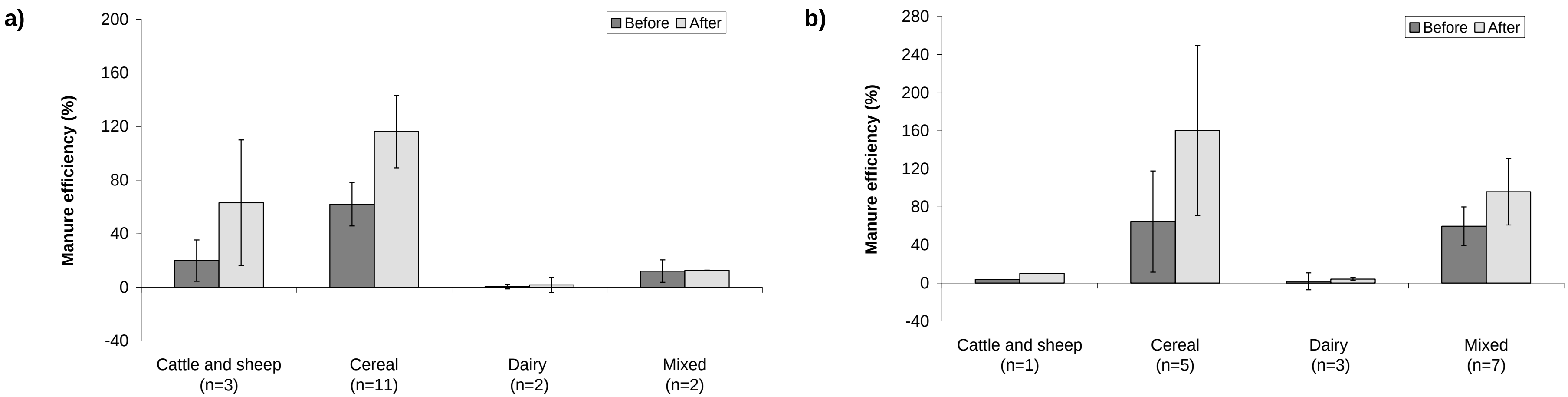


Figure 7-11: Manure efficiency (%) before and after the implementation of mitigation in a) MSA and b) EMEL. Mean and SE shown with sample size in parentheses. Differences between years not significant. See Table 7-11 for full results of statistical analysis.

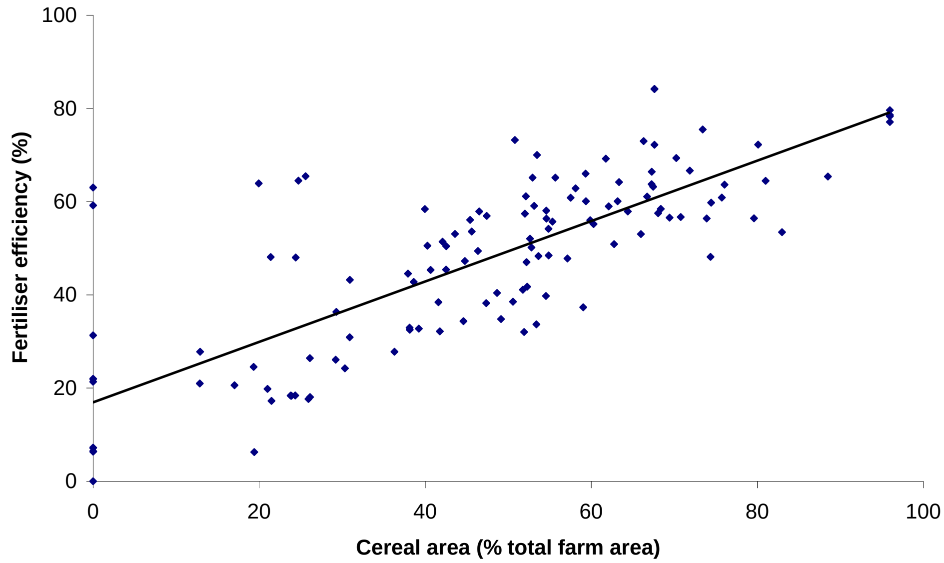


Figure 7-12: Relationship between cereal area (% total farm area) and fertiliser efficiency (%). R^2=0.620, y=0.6379x + 17.532. P<0.001

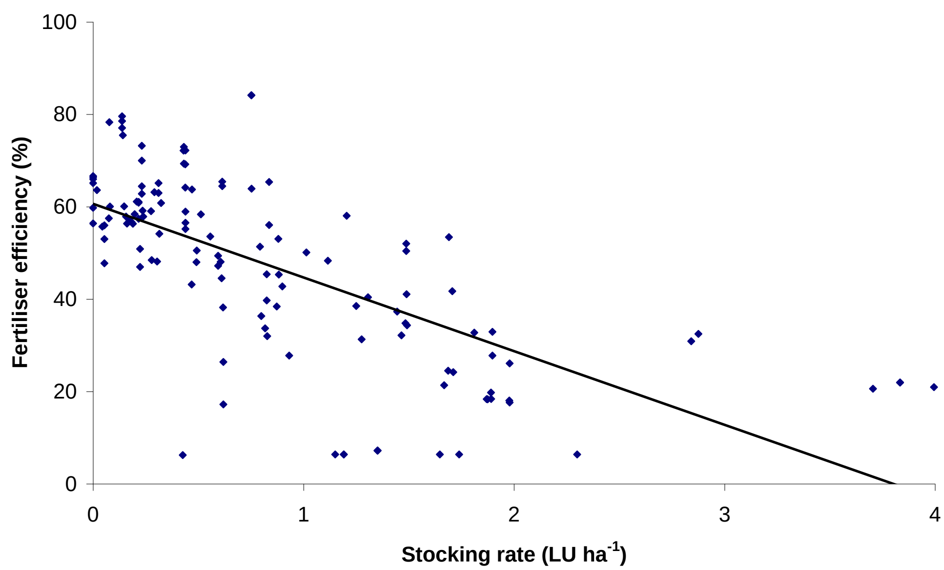


Figure 7-13: Relationship between stocking rate (LU ha^{-1}) and fertiliser efficiency (%). R^2=0.456, y=-15.952x + 60.637, p<0.001.9

Differences in efficiency before and after mitigation were not significant in MSA however it is worth noting the 47% increase in total efficiency observed on MSA cattle and sheep farms (Figure 7-10). Improvements stemmed from a 35.2% reduction in fertiliser inputs coupled with a 374.4% increase in crop export (Figure 6-3). ‘Hybrid’ farm budgets (see chapter 6) confirmed a reduction in fertiliser occurred despite changes in cropping. In addition, crop export increased at proportionally larger rate than a corresponding increase in arable area and reduction in stocking rates (Table 6-6). Manure efficiency also increased considerably however wide variability in results post mitigation meant the change was again not significant (Figure 7-11). Improvements stemmed from higher manure removal (Figure 7-14). The manure N balance remained relatively unchanged suggesting the improvement in manure efficiency was fuelled by an improvement in manure utilisation not a reduction in nutrient use.

MSA cereal farms

Cereal farms returned the highest total efficiency averaging 73.5% (Figure 7-10) . High crop export offset large inputs confirming the efficiency of arable production (Figure 6-3). Large cereal areas and low crop retention (due to low stocking rates) meant fertiliser efficiency was also high averaging 64.7%. High total outputs (removal) and small manure N balances meant manure efficiencies were, alike to total and fertiliser efficiency, highest on cereal farms averaging 75.6% (Figure 7-11 and Figure 7-14).

Following the adoption of mitigation total efficiency on cereal farms showed little change with improvements averaging 5.2% (Figure 7-10). Improvements in manure efficiency were more than ten times larger at 54.2% but remained non significant due to wide variability in results (Figure 7-11). Low manure N balances on cereal farms meant manure efficiency was sensitivity to changes in manure removal. Having said this, increased manure removal accompanied by little change in the manure N balance points to improved manure utilisation (Figure 7-14). However high manure N utilisation was also observed in 2006 questioning whether improvements were linked to mitigation.

Table 7-11: Results of ANOVA analysis for manure efficiency and associated inputs and outputs. See Figure 6-3 and Figure 6-4 for feed, fertiliser and manure inputs / outputs before and after mitigation. See Figure 7-14 for excreta and estimated manure removal before and after mitigation.

Catchment	Farm type	Feed	Feed eff.	Feed removal	Fertiliser	Crop retention	Fertiliser eff.	Est. Fertiliser removal	Manure In	Manure out	Excreta	Est. Manure removal	Manure efficiency
MSA	Farm type	P<0.001	P<0.001	P<0.001	P<0.001	P<0.001	P<0.001	P<0.001		P<0.001	P<0.001		P<0.001
	Year												
	Farm type x year												
EMEL	Farm type	P<0.001	P<0.05	P<0.001	P<0.001	P<0.001	P<0.001	P<0.001			P<0.001	P<0.001	P<0.001
	Year												
	Farm type x year												
MSA+EMEL	Catchment	P<0.001		P<0.001				P=0.051		P<0.05	P<0.001		
	Farm type x catch.	P<0.001		P<0.01	P<0.05		P<0.001			P<0.05	P<0.001		
	Farm type x catch. x year												

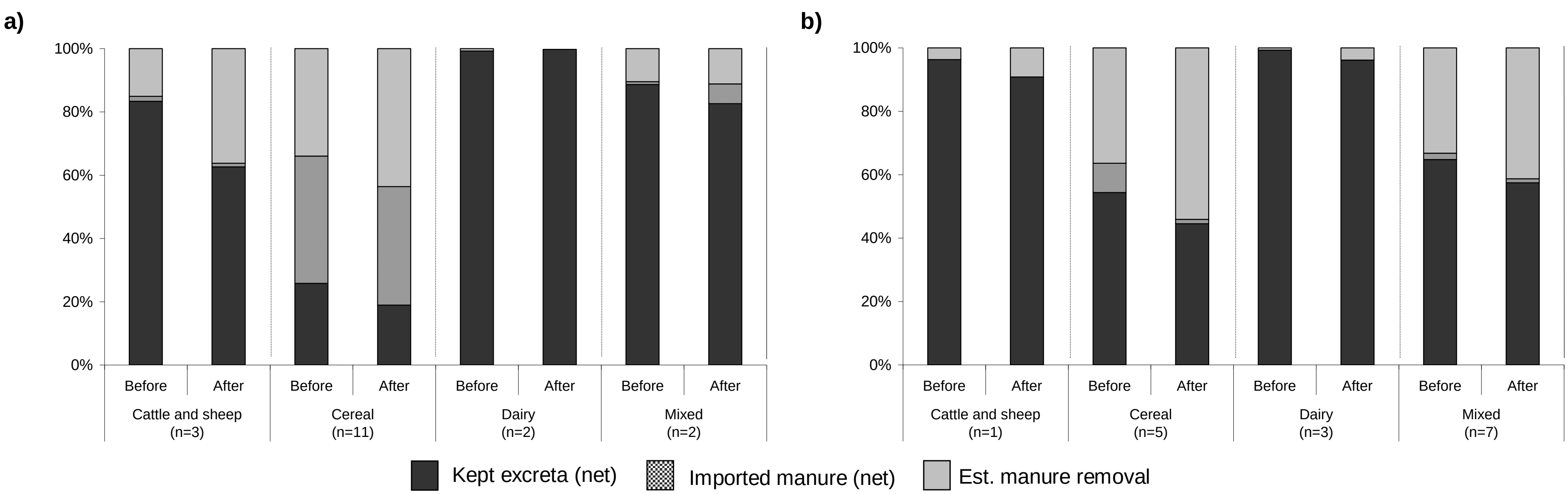


Figure 7-14: Inputs (excreta and imported manure N) and outputs (est. manure removal N) in a)MSA and b) EMEL before and after mitigation. Ammonia losses accounted for from excreta and imported manure. Kept excreta refers to total excreta minus exported excreta. Estimated manure removal refers to total removal minus removal attributed to feed and fertiliser. See Table 7-11 for results of statistical analysis.

MSA Dairy farms

Dairy farms represented the least efficient farm type in both catchments with total efficiency averaging 27.4% in MSA (Figure 7-10). Inputs were approximately three times larger than outputs (Figure 6-3). Fertiliser efficiencies on dairy farms were significantly lower in MSA than EMEL (19.74 vs. 27.22%) a result of higher crop retention on MSA due to higher stocking rates (Table 6-6 and Table 7-11). High excreta inputs combined with low manure output meant manure efficiencies were also low, averaging just 0.84% in MSA (Figure 7-11 and Figure 7-14).

Total efficiency decreased on dairy farms but not significantly (Figure 7-10). Reductions in efficiency were driven by increased feed which concealed a reduction in fertiliser use (Figure 6-3) . Hybrid surpluses (chapter 6) suggested cropping patterns after mitigation would demand more fertiliser than before mitigation and so despite the reduction being non significant, fertiliser use may have benefitted from mitigation. With reductions in fertiliser use counteracted by increased feed, dairy farm results demonstrate the need to assess change at the farm scale. Improvements in fertiliser use are encouraging, but need to be supported by good nutrient management in other areas of the farm. Modest improvements in manure efficiency were observed although differences before and after mitigation were not significant (Figure 7-11).

MSA Mixed farms

Total efficiency averaged 36.3%, 23.8% lower than in EMEL due to larger livestock / dairy enterprises and higher stocking rates which are inherently less efficient than crop production (Figure 7-10). Fertiliser efficiency was also lower in MSA a result of higher crop retention to support livestock (35.7 vs. 47.4%). Following the adoption of mitigation changes in total and manure efficiency were both small and insignificant (Figure 7-10 and Figure 7-11). However the relative stability in efficiency concealed proportional increases in both inputs (mainly feed) and outputs (crop and livestock products) (Figure 6-3). Changes in farm structure offer little explanation for the increasingly high input – high output production with increased feed and animal products contradicting a reduction in stocking rates (Table6-6). Although not significant, increased feed concealed a reduction in fertiliser which coupled with cropping conducive to higher fertiliser inputs suggests a positive response to mitigation with respect to fertiliser use. However as mentioned in chapter 6, increased feed meant the benefit of improved fertiliser management did not extend to improved total efficiency suggesting mitigation failed to have a whole farm impact and did not address inefficiencies in feed management.

EMEL cattle and sheep

In contrast to MSA cattle and sheep farms in EMEL returned the highest total efficiencies, averaging 96.3% (Figure 7-10). However results were highly variable meaning differences between catchments were not significant. Low inputs and outputs meant small nutrient deficits could equate to large relative differences generating large efficiencies (Figure 6-4). In contrast fertiliser and manure efficiencies were generally lower in EMEL than MSA (14.8 vs. 7.2% and 30.7 vs. 5.4% in MSA vs. EMEL for fertiliser and manure efficiency respectively) (Figure 7-11 – manure efficiency only).

A large and significant increase in total efficiency was observed after mitigation was adopted with average results increasing by 271.6% (Figure 7-10). Improvements were driven almost entirely by a reduction in fertiliser, decreasing to 0kg N ha^{-1} in the mitigation year (Figure 6-4). However the majority of this reduction occurred prior to the implementation of mitigation. Improvements in manure efficiency were insignificant and a reduction in excreta linked to falling stocking rates (Table 6-6, Figure 7-11, and Figure 7-14).

EMEL cereal farms

The efficiency (total, fertiliser and manure) of EMEL cereal farms was consistently high and similar to that in MSA (Figures 7-10 and 7-11). This stemmed from high crop export and low manure N balances (Figure 6-4 and Figure 7-14). Following the adoption of mitigation a significant increase in total efficiency was observed (Figure 7-10). Increased crop export meant average total efficiency rose by 9.5% (Figure 6-4). Although 2008 cropping was conducive to lower fertiliser use (see chapter 6), similar cropping in 2006 and 2008, but lower fertiliser in 2008, suggests more efficient nutrient use in the latter. Similar to MSA large improvements in manure efficiency were observed. However variability between years and farms was high with low manure balances highly sensitive to small changes in manure removal. As a result the improvements in manure efficiency were not significant (Figure 7-11).

EMEL dairy farms

Similar to MSA dairy farms, total efficiency, fertiliser efficiency and manure efficiency on dairy farms were low averaging 26.8%, 27.2% and 0.52% (Figure 7-10 and

Figure 7-11). However in contrast to MSA a significant improvement in total efficiency was observed on EMEL dairy farms post mitigation. Increased crop export led to a 7.2% increase in total efficiency (Figure 7-10 and Figure 6-4). Although fertiliser inputs also increased, this was counteracted by a reduction in feed. However a corresponding reductions in stocking (Table 6-6) and crop retention observed across all year suggests a shift towards crop production which in accordance with the relationships shown in Figure 7-12 and Figure 7-13 is conducive to higher efficiency. Improvements in manure efficiency were small and insignificant (Figure 7-11). Despite considerable scope for improved manure use on dairy farms, large manure N balances meant small increases in manure removal and a reduction in excreta had little effect on efficiency (Figure 7-14).

EMEL mixed farms

Differences in efficiency between catchments were not significant, however average total, fertiliser and manure efficiencies were considerably higher in EMEL than MSA (Figure 7-10 and Figure 7-11). This stemmed from lower stocking rates and larger arable enterprises on EMEL mixed farms (Figure 3-9). Mixed farms did however continue to represent an intermediate position relative to other farm types as observed in MSA. Following the adoption of mitigation total efficiency increased by 16.3% (Figure 7-10). Improvements were predominately driven by a large reduction in feed inputs, supplemented by increased crop export and reduced fertiliser inputs (Figure 6-4). Changes in farm structure offer little explanation for observed improvements with cereal area and stocking rates relatively unchanged (Table 6-6). However, efficiency increased progressively across all years and while reductions in feed are encouraging, none of the mitigation methods employed targeted feed management directly. Manure efficiency increased by 36.6% however changes before and after mitigation were not significant (Figure 7-11). Increased manure removal (Figure 7-14), suggests improved manure utilisation but alike to EMEL cereal farms, results displayed wide variability.

7.3.2 Sensitivity to mitigation level (GAP vs. EGAP) and manure management plans (MMP)

The level of farm scale mitigation (GAP vs. EGAP) had no significant effect on improvements in total or manure efficiency post mitigation years. This was despite larger improvements in total and manure efficiency corresponding with higher level mitigation on all but cattle and sheep farms (total efficiency) – data not shown. Manure management plans (MMPs) had a significant effect on total efficiency, however positive responses were limited to cattle and sheep and dairy farms (Figure 7-15). MMPs corresponded with larger improvements in manure efficiency on all but cereal farms but differences in improvements on farms with / without an MMP were not significant (Figure 7-16).

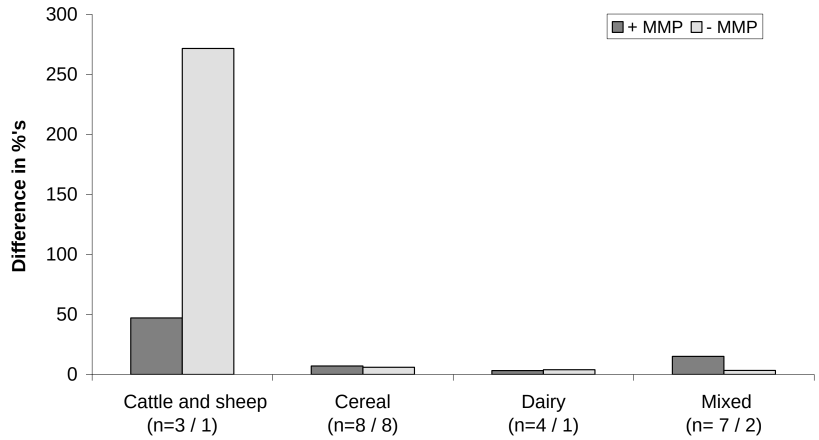


Figure 7-15: Effect of manure management plans (MMP) on total efficiency. Figure shows average change in efficiency (%) between before and after results for MSA and EMEL combined. N refers to number of farms with (+) / without (-) MMP agreements respectively. Differences +/- MMP significant to p<0.05. year to p<0.01 and farm type to p<0.001. Significant interaction between MMP, farm type and year (p<0.001).

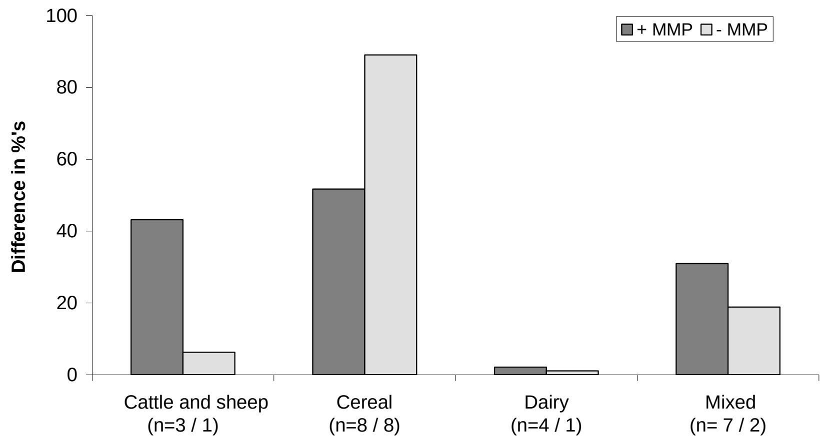


Figure 7-16: Effect of manure management plans (MMP) on manure efficiency. Figure shows average change in efficiency (%) between before and after results for MSA and EMEL combined. N refers to number of farms with (+) / without (-) MMP agreements respectively. Differences +/- MMP not significant and no significant interactions involving MMP.

7.3.3 Relationships between 'efficiency' and farmgate surpluses

Efficiency results were compared to farmgate surpluses (chapter 6) to investigate whether 'efficiency' is more sensitive to mitigation than surpluses, and whether the calculation of input specific efficiencies expose responses to mitigation not observed in whole farm calculations. Comparisons were also made to increase confidence in the reliability of efficiency results; the two approaches are fundamentally different (one representing the balance of inputs and outputs, the other equating the relative size of inputs and outputs) but include very similar components.

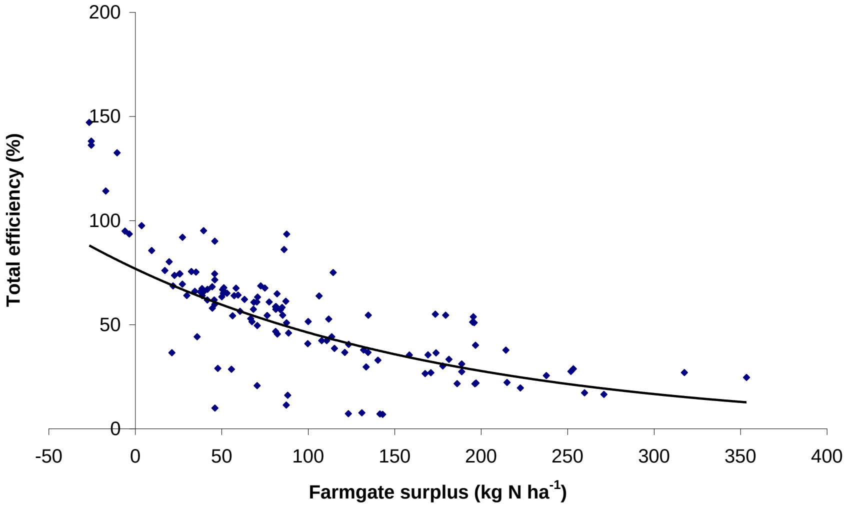


Figure 7-17: Relationship between farmgate surplus (kg N ha^{-1}) and efficiency (%) across all farms. R^2 =0.394, Y=76.87e -0.0051x p<0.001. See Chapter 6 for details of farmgate surplus calculations.

A significant inverse relationship was observed between farmgate surpluses and total efficiency (Figure 7-17). Farms operating with large surpluses were generally less efficient than those with small surpluses. Maximum efficiency and smallest surpluses were observed on cereal farms in MSA (cattle and sheep farms in EMEL), whilst lowest efficiency and highest surpluses were observed on dairy farms. Mixed farms occupied an intermediate position in both approaches. Increased scatter at lower surpluses reflects the different levels of production and efficiency of cattle and sheep and cereal farms; both farm types returned small surpluses however efficiency was much lower on cattle and sheep farms confirming the low input – output production previously suggested (section 7.3.1). The presence of farm type specific relationships (Figure 7-18) supports the farm type differences observed in both farmgate and efficiency results. However it also demonstrates that for a given farm type, and thus within the inherent nutrient utilisation constraints associated with that farm type, there is an opportunity on most farms to reduce surpluses through improved efficiency.

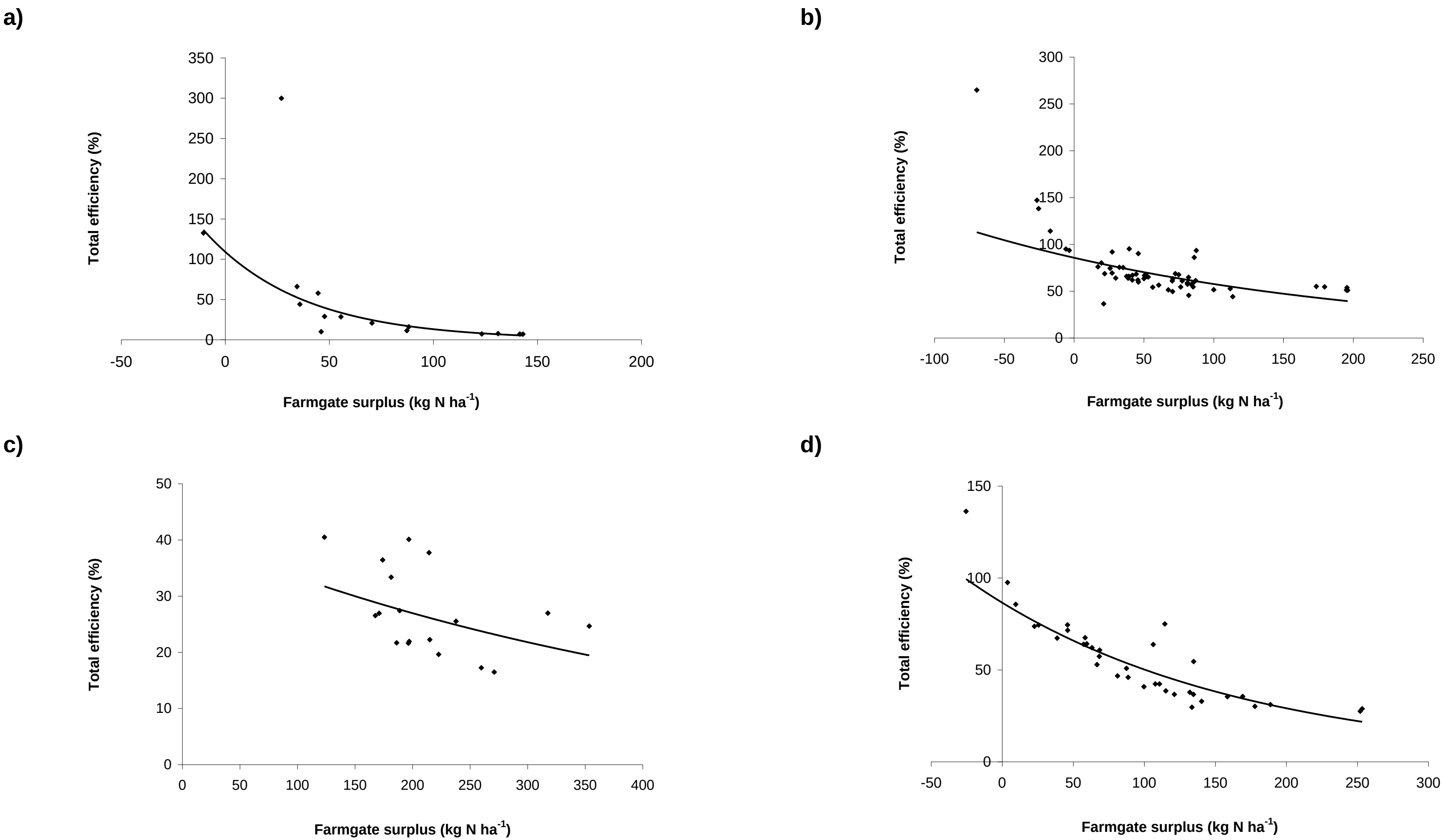


Figure 7-18: Relationships between farmgate surpluses (kg N ha^{-1}) and efficiency (%) on a) cattle and sheep farms: R^2=0.724 y=109.13e -0.0212x, p=0.026 b) cereal farms: R^2=0.439, y=85.659e -0.004x, p<0.001 c) dairy farms: R^2= 0.188, y=41.255e -0.0021x p=0.068 and d) mixed farms: R^2=0.7961, y=86.543e -0.0054x, p<0.001

Comparison of efficiency and surpluses expose differences in production intensity between MSA and EMEL. Surpluses on MSA dairies were significantly larger than in EMEL despite similar efficiencies (Figure 6-2 and Figure 7-10), a result of higher inputs and outputs stemming from more intensive production on MSA dairies; where equally efficient systems are scaled up, the quantity of inefficiently used nutrients i.e. the surplus, increases. Surpluses were also significantly larger on cereal farms in MSA than EMEL, however in contrast to dairy farms, MSA cereal farms also tended to be more efficient. If production in EMEL were to be scaled up to the same level as in MSA, surpluses would exceed those observed in MSA. On cattle and sheep farms surpluses and efficiency were both higher in EMEL suggesting production was more intense in EMEL.

7.3.3.1 Responses to mitigation

Following the implementation of mitigation, total efficiency and surpluses improved on a very similar proportion of farms (76.5 vs. 79.4% respectively). Improvements in efficiency coincided almost exactly with improvement in surpluses with just one farm improving its efficiency only, and two farms reducing only their surplus. Improvements in manure efficiency were observed on a slightly larger proportion of farms, increasing on 84.4% of farms. On a catchment – farm type basis, significant improvements were observed in EMEL only for both budgets and efficiency. Largest improvements were observed on cattle and sheep farms via both methods (Figure 6-2 and Figure 7-10). Improvements in total efficiency, manure efficiency and surpluses tended to be larger where higher levels of mitigation were adopted however differences were not significant.

In MSA comparison of surplus and efficiency results revealed an increase in production intensity on mixed farms (section 7.3.1). While a proportional increase in inputs and outputs left efficiency largely unchanged, higher input – output production increased surpluses. This result highlights the fundamental difference between balance and efficiency methodologies, that productivity is respected, and farms not disadvantaged for increasing production intensity if nutrient utilisation is maintained. However with this comes larger surpluses, posing a greater risk of N loss to the environment.

The extent to which change was attributed to mitigation was similar between the two approaches, reflecting similar inputs and outputs and therefore similar sensitivity to structural and environmental change. Both methods highlighted the need for mitigation which addresses nutrient use across the whole farm, and the benefit of evaluating mitigation at the farm scale to capture indirect responses and compensatory behaviour, for example changes in feeding regimes following changes in fertiliser applications to grass. An absence of manure related responses to mitigation in farmgate surpluses and total efficiencies may expose short comings of farm scale assessment methods for the evaluation of manure use and thus the benefit of adopting a manure specific indicator. However improvements in manure efficiency were themselves not significant. Differences in sensitivity to manure based mitigation may indicate methodological problems and over sensitivity of manure efficiency to changes in the manure balance and fertiliser efficiency

7.4 Discussion

7.4.1 Mitigation sensitivity

Following the implementation of mitigation, total efficiency increased on 77% of farms and manure efficiency on 85% of farms. However only in EMEL and for total efficiency only were improvements significant. Increasing from 53 to 72% (averaged across all farms) improvements were in line with those observed in Denmark between 1979/1980 and 2003/4 following the implementation of National Action Plans aimed at reducing nitrate leaching by 50% (Kyllansbaek and Hansen, 2007). However given the extent of mitigation connected with Danish Action Plans which included sub optimal fertiliser, manure N limits and extensive growth of cover crops, and the contrasting time scales over which results have been obtained, improvements are unlikely to be equally attributable to mitigation.

On a farm type basis improvements in total efficiency were generally larger than those reported in the literature. For example, the 47.1 / 271.6% increase on cattle and sheep farms and 5.2/9.5% increase on cereal farms (MSA/EMEL) exceeded improvements on Flemish farms which averaged 8% and 2% on cattle and sheep and cereal farms respectively following the effort to improve nutrient management (Verbruggen et al., 2005). Only on dairy farms were improvements in EMEL (efficiency decreased on MSA dairy farms) of a similar magnitude to those observed elsewhere. Improvements observed in the Netherlands and Belgium (Verbruggen et al., 2005; Nevens et al., 2006; Van Wepern, 2009) following reductions in fertiliser use, improved manure management, improved feed management, increased manure storage and the growth of cover crops were in line with the 7% improvement observed in EMEL. Irrespective of mitigation, total efficiency results were in broad agreement with the literature, confirming the inherent inefficiencies attached to livestock production and increasing confidence that all major nutrient fluxes were retained in the efficiency methodology (Table 7-12).

Table 7-12: Comparison of MSA and EMEL farm efficiencies (%) with results in the literature. Full details of contributing references can be found in table A-6 in the appendix.

Farm type	Observed total efficiency (%)		Efficiencies reported in literature (%)		
	MSA	EMEL	Average	Max	Min
Cattle and sheep	33	96	29	38	19
Cereal	73	67	62	73	51
Dairy	27	27	28	52	14
Mixed	36	60	51	66	35

Despite improvements on a large proportion of farms, total efficiency in EMEL and MSA remained lower than observed on demonstration farms such as the De Marke dairy farm in the Netherlands which operates at efficiencies in excess of 30% (Aarts et al., 2000). However such high efficiencies requires considerable changes to farm structure and management including reduced stocking rates, reduced grazing seasons and daily grazing, reduced grass area, and increased maize production. The financial rewards for achieving higher efficiency and the flat rate payments for additional mitigation methods were not have been sufficient to encourage this level of change on MSA / EMEL farms. Indeed improvements on EGAP farms were not significantly larger than those on GAP, however higher level mitigation was generally implemented at field scale, reducing apparent impact when evaluated at the farm scale.

With respect to manure efficiency, only in Germany has a similar manure efficiency approach been adopted (also using a modified version of the original German methodology). Following the adoption of mitigation similar to WAgriCo mitigation, only 20% of farms improved their manure efficiency (WAgriCo, 2008). Differences in the extent of improvements may stem from methodological differences, differences in the sensitivity to external factors affecting nutrient use and differences in the degree of change arising from mitigation due to different ‘baseline’ situations and levels of farmer support. Despite a tendency towards improved manure efficiency post mitigation in EMEL and MSA, interpretation of total efficiency results did not expose any improvements in manure management; however this may confirm the need to evaluate manure efficiency separately.

As acknowledged in chapters 5 and 6, external factors affecting nutrient use and offtake coupled with annual changes in farm structure and cropping meant improvements in total efficiency could not be directly attributed to the implementation of mitigation. Results must therefore be interpreted relative to other confounding variables and inter-annual variation. Once changes in farm structure, cropping and annual variability were accounted for, only the improvements in total efficiency on cattle and sheep farms (both catchments) and cereal farms (EMEL only) appeared connected to the implementation of mitigation, although even then changes in fertiliser price and environmental conditions are likely to have impacted on nutrient use decisions and crop offtake. The improvements on MSA cereal farms can instead be linked to changes in cropping, whilst changes on EMEL dairy farms reflect increased emphasis on crop production. Although improvements were also observed on EMEL mixed farms, changes were largely driven by changes in feed, an aspect of nutrient use not addressed by the mitigation employed and thus stemming from other factors. (WAgriCo promoted fertiliser and manure based mitigation on the grounds that it would be more widely applicable and acceptable, and would address excessively high manure applications made locally). Conversely increased feed inputs counteracted reductions in fertiliser on MSA dairy and mixed farms highlighting the need for mitigation to address whole farm nutrient management. Given the benefit of balanced rations and reliance on home grown feed on farm efficiency noted by Powell et al., (2009), greater mitigation driven change may have been observed had mitigation also targeted feed management.

Limited mitigation induced change and short evaluation timescales are likely to have contributed to the lack of positive responses to mitigation in MSA and EMEL, however previous studies have highlighted the sensitivity of efficiency to mitigation. Kohn et al., 1997 observed an almost like for like increase in feed utilisation and farm efficiency, confirming the theoretical sensitivity of farm efficiency to changes in livestock diet and management. This is in agreement with Powell et al., (2009) who acknowledged the benefit of improved feed management on dairy farm efficiency.

Kohn et al., (1997) also observed sensitivity to improvements in crop nutrient utilisation, equating a 50% improvement in crop uptake to a 59% increase in farm efficiency. This is supported by Bleken (2009) and Van Bruchem et al., (1999), who both acknowledged the sensitivity of farm efficiency to changes in the efficiency of soil – plant transfers. However opportunities to improve efficiency via improved manure retention and utilisation in particular were considered low, a finding which contradicts the larger improvements observed in manure than total efficiency in MSA / EMEL. However direct comparison may not be valid given that improvements have not been fully attributed to mitigation. While sensitivity to mitigation and changes in individual pathways is encouraging, it does not represent direct evaluation of mitigation method effectiveness. Longer term commitments to mitigation implementation, greater mitigation induced change and evaluations conducted over longer timescales are therefore required to explore the usefulness of efficiency further.

7.4.2 Methodological discussions

Evaluating nutrient use efficiency on a total N, fertiliser N and manure N basis provided an opportunity to address conflicting opinions regarding evaluation scale. Calculation of total efficiency ensured farm components were not treated in isolation avoiding localised improvements at the expense of other components (Van Bruchem et al., 1999). For example improvements in fertiliser use on mixed and dairy farms were counteracted by increased feed imports - only by calculating efficiency at the farm scale was the net effect of these conflicting changes fully exposed. However complementing farm scale evaluation with separate calculations of manure and fertiliser efficiency respected the need for more detailed, transfer specific auditing techniques (Bleken, 2009) and the importance of farm management and strategic structure which affect feed import and crop export fractions and thus demand quantification of efficiency along each nutrient transfer (Schroder et al., 2003). However unlike previous suggestions of sub-compartmental calculation, conversion coefficients for each transfer and differentiation between trophic levels (plant and animal pathways), differentiation between fertiliser and manure N allowed the efficiency of complete pathways to be quantified irrespective of whether inputs led to crop or livestock produce, whilst still providing a more detailed representation. Given the importance of excreta and manure handling, and the herd – field transfer acknowledged by Bleken et al., (2005) and Borsting et al., (2003), isolation of manure N and the retention of links between field and herd via grazing and excreta are especially useful. With respect to mitigation, methods tend to target specific N sources making corresponding N source specific calculations well suited to their evaluation.

While the theoretical benefits of calculating total, manure and fertiliser efficiency are clear, complexity and system simplifications associated with fertiliser and manure efficiency calculations mean their value must be balanced against heightened methodological uncertainty. However the novelty of the approach means similar approaches are uncommon in the literature reducing opportunities to highlight and address areas of particular concern. Comparison of corresponding total, manure and fertiliser efficiencies was instead used to highlight areas of uncertainty.

Based solely on inputs and output and free from internal transfers and efficiencies, total efficiency characterises nutrient use less explicitly but yields more transparent and interpretable results. Manure and fertiliser efficiency are instead interlinked with uncertainties in one affecting the reliability of the other, and differences in derivation affecting the reliability of comparisons. Manure efficiency was generally higher than fertiliser efficiency, an unexpected result given that manure management presents greater opportunities for improvement and manure N has a smaller readily available N component than fertiliser. Conversely negative manure removal and efficiency was observed on some farms suggesting fertiliser efficiencies were in some instances were too high. Manure and fertiliser efficiency calculations require refinement to ensure results are comparable and consistent. Where manure N balances were low, for example on cereal farms, manure efficiency was also highly sensitive to changes in removal, returning very high and very low efficiencies. While this is unavoidable where inputs and outputs are divided, the instability of results means manure efficiencies should be interpreted with respect to manure N balances. Similarly, very large total efficiencies were calculated for cattle and sheep farms in EMEL. While this is more likely a result of advanced purchasing of fertiliser / feed, small negative balances equate to very large efficiencies resulting in misleading results.

With regard to mitigation, methodological differences between fertiliser and manure efficiency calculations confounded the interpretation of improvements. Although offering a more detailed representation of actual fertiliser utilisation than a single default value, the sensitivity of fertiliser efficiency to changes in cropping meant reductions in fertiliser use did not fully transpire in manure efficiency, limiting the extent to which improved accounting of manure was apparent in manure efficiency. Results were also limited by a lack of mitigation sensitivity attached to fertiliser efficiency. Fertiliser efficiency was used only as a means to calculate manure efficiency and was not interpreted in its own right, however it was possible that fertiliser use improved but manure use was unchanged, demanding sensitivity to mitigation in both indicators. In the absence of such sensitivity, reduced fertiliser inputs increased manure removal and were therefore linked to better manure not fertiliser management. This would explain why improvements in total efficiency were not attributed to improved manure management despite improvements in manure efficiency. An absence of excreta in total efficiency calculations is also likely to have reduced sensitivity to manure mitigation, supporting the calculation of manure and fertiliser efficiency especially where reliance on manure imports and exports is low.

Calculating total, fertiliser and manure efficiency also aimed to allow differentiation between genuine improvements in efficiency and shifts in reliance between fertiliser and manure N; the higher crop availability of fertiliser N means farms reliant on fertiliser N tend to have higher total efficiencies than those utilising predominately manure N. Where a substantial increase in fertiliser N and decrease in manure N was observed, (fertiliser increased from 20708 to 23369kg N while manure decreased from 20155 to 12657kg N) both total and manure efficiency increased suggesting structural change was effectively accounted for in results. Instances of increased reliance on manure N were less evident, however where manure use did appear to increase at the expense of fertiliser, total efficiency still tended to increase, a result of disproportional increases in feed and excreta (only feed inputs are included in total efficiency calculations) and increased removal. Changes in manure efficiency were farm specific, also reflecting the balance of feed and excreta N and changes in removal; manure efficiency increased where increases in removal exceeded that of manure, and decreased where excreta increased more than feed. Methodological differences between total and manure efficiency calculation coupled with changes in cropping, stocking and mitigation make it difficult to assess the extent to which structural change was accounted for in these examples. Evaluations would benefit from calculation of fertiliser efficiency from actual farm data to increase the completeness of each farm's evaluation and to provide confirmation that where manure use increased, increased total efficiency stemmed from improved fertiliser efficiency not methodological differences. However with few farms displaying substantial shifts in manure and fertiliser N reliance, MSA and EMEL evaluations were little affected by this.

7.4.3 Comparison of 'efficiency' with farm gate budgets

Comparison of surpluses and total efficiency results highlighted differences in sensitivity to farmgate budgets and efficiency to production intensity. While surpluses were similar on cereal and cattle and sheep farms, efficiency was much higher on cereal farms. Lower surpluses on cattle and sheep farms stemmed from low input – low output production not efficient production. Comparison of changes before and after mitigation highlighted differences in sensitivity to changes in production intensity and the implications of this on evaluations of mitigation effectiveness. Intensification of production on MSA mixed farms led to increased surpluses but had little effect in efficiency. Efficiency has the advantage of not penalising farms which undergo strategic change at the same time as adopting mitigation. However larger surpluses pose greater risk the environment despite efficiency remaining constant. With mitigation aiming to reduce losses, maximum benefit may be achieved where the two indicators are used together to ensure increased efficiency does not have a detrimental effect in nutrient surpluses.

Significant inverse relationships between surpluses and efficiency highlighted opportunities for many farms to reduce their surplus through improved efficiency. The presence of farm type specific relationships confirm that improvements are possible within the inherent nutrient utilisation constraints associated with a particular farm type. While quantifying efficiency allows nutrient use to be compared against maximum achievable efficiencies, respecting the inefficiency of livestock feed conversion and lower availability of manure N than fertiliser N, these relationships demonstrate what is possible within a particular farm system whilst confirming both an economic and environmental gain. By moving up the trendline a twofold cost benefit is exposed. Lower surpluses reduce the loss of a valuable resource, whilst higher efficiency is likely to yield greater output. For example on cereal farms a 10% increase in efficiency, similar to that observed in EMEL, would result in a 30kg N ha^{-1} (50%) reduction in the farmgate surplus (Figure 7-19). An interesting next step would be to consider the cost savings (and cost benefits) associated with such changes, and quantify more explicitly the changes in management required to achieve these efficiencies. It must however be remembered that even within a given farm type variation in production intensity and relative arable – livestock enterprises exists. Indeed these variables explained much of the change in nutrient use during this short term project, the differences between mixed farms in EMEL and MSA and in part the range of efficiencies reported in the literature. However in contrast to the messages transpiring in other chapters these relationships transcend the issue of annual variability.

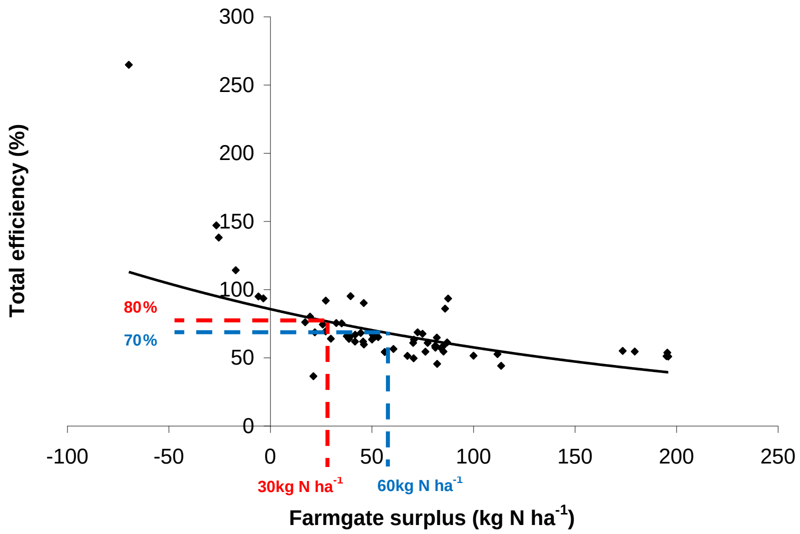


Figure 7-19: Example of how improved efficiency on cereal farms would reduce surpluses.

Dissemination of results within the WAgriCo arena demonstrated the high level of interest farmers have in the results and success of their peers and neighbours. By presenting results from across the catchment these relationships not only demonstrate opportunities for improvement but have the potential to drive improvements. Farmers appeared motivated by friendly competitiveness with their neighbours, wanting to have the most profitable farm but one that appears to be environmentally aware (farmers preferred mitigation which 'looked' good). Seeing neighbouring farms operating more efficiently is a good 'carrot' for driving change.

7.5 Conclusions / Further Work

Improvements in efficiency following the implementation of mitigation exposed sensitivity to changes in nutrient use, however annual variability in cropping, stocking, weather and prices meant changes were unlikely to be mitigation driven in most cases. Longer term evaluations, more extensive mitigation, and detailed understanding of why changes in nutrient occurred obtained through dialog with farmer are required to isolate the impact of mitigation from other variables and to assess in more detail the usefulness of efficiency as a means of mitigation evaluation. Efficiency results did however highlight the need for whole farm mitigation to address feed management and the benefit of evaluating mitigation at the farm scale to ensure the net effect of conflicting changes are captured.

Correspondence between efficiency and balance results confirmed inclusion of major nutrient fluxes within the efficiency methodology, however efficiency results exposed differences in productivity which were less evident in farm gate balance surpluses. Efficiency has the potential to evaluate mitigation independent of changes in production intensity, and allows current nutrient utilisation to be compared against maximum achievable values. However, with increasing production at a given efficiency comes larger surpluses, and with surpluses more directly linked to losses than efficiency, calculation of both surpluses and efficiency would benefit evaluations where substantial changes in also productivity occurs. Indeed data requirements are similar and little additional manipulation would be required.

Relationships between farmgate surpluses and efficiency highlighted opportunities for many farms to reduce their surplus through increased efficiency whilst acknowledging the nutrient utilisation limitations associated with different farm systems. For maximum farmer buy in and engagement, these relationships also demonstrate the cost benefit attached to higher efficiency through lower surpluses whilst dissemination of performance relative to peers acts an effective initiator of change. Coupled with calculation at scale of relevance to farmers and opportunities to highlight priority farms, efficiency (combined with farmgate surpluses) represents a useful for farmer engagement and the cost effective attainment of lower surpluses through improved efficiency.

Broad agreement between total, manure and fertiliser efficiency by farm type confirmed the modified methodology was generally robust and consistent across its different components. The consistency of results also confirmed that the methodological refinements undertaken (complete accounting of feed / excreta N, and more explicit representations of fertiliser and feed efficiency) were not at the detriment of output reliability. Manure efficiency provided a useful insight into the utilisation of own farm manure which is largely absent from total efficiency, offering more detailed evaluations of manure related mitigation at the farm scale. However methodological differences between total and manure efficiency, namely feed vs. excreta questioned the validity of these comparisons. In addition a lack of mitigation sensitivity attached to fertiliser efficiency meant improvements were potentially over attributed to improved manure utilisation. Increased sensitivity of fertiliser efficiency to mitigation would improve evaluations of manure mitigation, heighten sensitivity to fertiliser mitigation and aid assessments of the impact and consideration of structural change. With fertiliser efficiencies consistently lower than manure efficiencies, and instances of negative removal, further work is also required to ensure flows are balanced and methodologies consistent between the different components. In doing so comparisons between indicators would be more meaningful and confidence that indicator specific responses to mitigation were not simply a reflection of methodological differences increased. Results confirm that separate manure and fertiliser efficiency calculations are possible and have the potential to effectively complement farm scale evaluations of total efficiency.

8 General Discussions and Conclusions

8.1 Introduction

This chapter aims to compare measurement, budgets and efficiency as assessment methods based on the results presented in chapters 4 - 7. In the first instance measurement and budget results are compared to assess the links and limitations associated with different methods at different scales. Discussions then focus on the extent to which the assessment methods investigated meet the requirements of an effective assessment method with specific reference to the demands of the WFD (Table 8-1). On the basis of these discussions, recommendations will be made as to the way forward; which assessment methods are recommended (if it is felt such comments can be made from the results obtained), at what scale assessment should be conducted, what mitigation is recommended and how, based on the experiences from the WAgriCo project farmers are best engaged.

8.2 Linking scales and methods – a critique of results

Comparison of budget and measurement results highlighted limitations in the assumption that surpluses are an indicator of likely loss in the short term. Correspondence between field scale surpluses and loss was low (Table 8-2), however this could in part be attributed to difficulties in making like for like comparisons where measurement and budget approaches reflect activities over different timescales and at different points of the harvest year. Greater agreement between methods was therefore observed where crop rotations were also considered. For example autumn SMN for WWF was high because it typically followed WOSR which leaves large N residues. However PP measurements which continued through the April were much lower reflecting over winter uptake and immobilisation of N in plant material. WOSR surpluses were high due to high fertiliser inputs, however high N inputs were not captured in measurement because fertiliser applications were generally made after SMN and PP sampling. Early establishment of WOSR instead meant high N uptake over winter and lower losses.

For long term grass, measurement and surpluses were contradictory exposing the significance of mineralisation and nutrient uptake which are poorly captured by SMN and field scale budgets. Indeed budgets afford no consideration to the availability of surpluses for loss, ignoring temporal variability in release, demand and transport processes (Watson and Atkinson, 1999; Oborn et al., 2003).

Table 8-1: Requirements of an effective assessment method with reference to the demands of the Water Framework Directive (WFD). Requirements 5-8 are referred to as usability in the following discussions.

	Requirement	Relevance to the WFD
1	Provide evidence of a reduction in nutrient loadings, concentration or ecological functioning.	→ Waterbodies are required to reach good ecological and chemical status, demanding a reduction in nitrate loadings and concentration.
2	Demonstrate sensitivity to a wide range of mitigation methods and provide assessments of relevant detail	→ Assessment methods must be sensitive to policy relevant mitigation methods. → Assessment methods must identify which individual mitigation methods are most effective whilst also considering the cumulative response which impacts water bodies.
3	Provide assessments at a scale of relevance.	→ The WFD is implemented at the river basin scale and assessed in waterbodies which reflect management at the catchment scale. However nutrients are managed, applied and lost at the field / farm scale. → High risk areas must be located and targeted to ensure mitigation is cost effective.
4	Provide confirmation of mitigation impact over a suitable timescale.	→ Good status must be achieved by 2015. → Reporting cycles extend over 6 years → PofMs must be operational by 2012, 3 years before good status is required.
5	Respect data and resource availability.	→ Economic analysis and cost effectiveness underpins the WFD.
6	Be practical and suited to end users.	→ The WFD promotes co-operation with stakeholders and encourages public participation.
7	Be sensitive to environmental and agricultural conditions.	→ PofMs are tailored to environmental and agricultural conditions. Assessment methods must be equally sensitive to catchment specific conditions.
8	Be relevant from a control and legislative enforcement perspective.	→ Waterbodies must comply with the 11.3mg N l^{-1} drinking water standard. → Compliance is on a ‘one out all out’ basis. → Land managers must accept responsibility and be accountable.

More detailed investigations between surpluses and loss highlighted the sensitivity of PP – loss relationships to rainfall, soil type and land management. For example 2007 and 2008 PP concentrations were more strongly related than PP and field surpluses suggesting that in the short term soil properties have more impact on leaching than nutrient management and cropping. Relationships were no more prevalent where analyses were conducted across the entire sampling period confirming four year evaluations are too short to account for fluctuations in soil N (e.g. the longer release of N from manures) and weather. Indeed positive relationships between surpluses and measured loss are generally restricted to longer term studies (>8years) (e.g. Koraeth and Eltun, 2000; Sieling and Kage, 2006; and Salo and Turtola, 2006), and relationships more significant where factors such as rainfall and land use are explicitly accounted for (e.g. Bechmann et al., 1998; Korsaeth and Eltun, 2000; Lord et al., 2002).

Table 8-2: Comparing surpluses (kg N ha^{-1}) and measured loss (kg N ha^{-1} / mg N l^{-1}) at the field scale. Figures in brackets denote relative crop ranks.

Current Crop	Autumn SMN [a]		PP load [b]		PP conc.[a]		Soil surface surplus [b]	
	kg N ha^{-1}		kg N ha^{-1}		mg N l^{-1}		kg N ha^{-1}	
Grass	152.7	*(2)*	84.8	*(7)*	33.4	*(7)*	117.6	*(1)*
Maize	120.1	*(5)*	174.9	*(4)*	88.4	*(2)*	26.5	*(7)*
SBM	107.5	*(7)*	127.7	*(5)*	65.3	*(3)*	38.0	*(5)*
WBF	119.4	*(6)*	233.1	*(2)*	62.4	*(4)*	56.7	*(4)*
WO	123.8	*(4)*	290.3	*(1)*	88.6	*(1)*	30.1	*(6)*
WOSR	131.0	*(3)*	110.2	*(6)*	38.1	*(6)*	117.1	*(2)*
WWF	164.0	*(1)*	196.1	*(3)*	62.1	*(5)*	90.5	*(3)*

[a] Differences between crops not significant to $p<0.05$

[b] Differences between crops significant to $p<0.05$

Table 8-3: Summary of responses to mitigation by measurement and budget assessment methods. ✓ denotes a positive (and significant) response to mitigation, ✗ denotes a negative response to mitigation. No symbol denotes non significant response / where statistical analysis was not appropriate.

ASSESSMENT METHODD		SCALE	RESPONSE TO MITIGATION
Measurement	*SMN*	Field	✓ Significant reduction in EMEL and MSA. ✓Significant reduction in over winter loss from spring cropped fields with cover crops vs. spring cropping fields left bare. ✓Significantly less over winter loss from grass than winter crops and stubble
	PP	Field	✗ Significant increase (by crop type) in EMEL and MSA Tendency (but NS result) towards lower leachate concentration and leached load (maize only) where spring crops preceded by cover crop.
	Groundwater quality	Catchment	Improvements at 44% and 72% sites in MSA and EMEL respectively. Improvement in average concentration and % exceedances of DWS in EMEL only. Reduction in maximum concentration in MSA only
Nutrient budgets	*Soil surface*	Field	✓Significant reduction in soil surface surpluses in EMEL only. Significant reduction in fertiliser inputs in MSA only ✓Significant crop specific effect of MMPs on reductions in soil surface surpluses and fertiliser inputs ✓Significantly lower surpluses and manure inputs where manure application delayed from late autumn to spring (1 farm only) Tendency (but NS result) towards lower surpluses spring crops preceded by cover crop
		Catchment	25.1 and 28.1% reduction in MSA / EMEL surplus
	Farm gate	Farm	Improvements on 79.4% of farms (77.8% MSA farms and 81.3% EMEL farms) ✓Significant reductions in EMEL (by farm type). NS but larger improvements on EGAP farms than GAP farms
		Catchment	8.1 and 8.7% reduction in MSA / EMEL surplus
	Efficiency	Farm	Improvements on 76.5% of farms (77.8% MSA farms and 75% EMEL farms) ✓Significant increases in EMEL (by farm type). NS but larger improvements on EGAP farms than GAP farms

Greater correspondence between surpluses and measured loss was observed on a response to mitigation basis than a nutrient characterisation (crop type / farm type / catchment differences) (Table 8-3)., suggesting the presence of an absolute relationship between surpluses and loss is not a pre-requisite to effective budget based evaluations of mitigation effectiveness However in the absence of longer timescales, there is considerably uncertainty as to whether improvements were mitigation driven. Indeed the factors governing nutrient use and loss were conducive to lower losses and surpluses post mitigation. Inconsistencies between surplus and measurement responses on a site specific basis reduce confidence that improvements were indeed mitigation driven – around 50% of sites displayed conflicting measurement and budget responses. However site specific comparisons were mostly of a cross scale nature and therefore sensitive to spatial variability, differences in spatial extents, and the scale dependency of inputs and loss processes on surplus – loss relationships. Furthermore, geographical proximity between fields and WQ sample sites does not guarantee connectivity between sources of available N and groundwater. With respect to relationships between farm surpluses and WQ measurement, comparison of field and farm scale surpluses confirmed surpluses are unevenly distributed across the farm. In addition, long transit times associated with groundwater responses means relationships between surpluses and catchment scale WQ are likely to encounter substantial time lags.

Comparison of field and farm scale budgets highlighted sensitivity to the scale and choice of nutrient budget methodology with soil surface and farmgate surpluses showing little correspondence both spatially (e.g. Figure 8-1)) and through regression analyses. Annual field scale surpluses poorly captured crop rotations, instead displaying hotspots associated with particular crops (e.g. WOSR). However with intra-farm variability in surpluses exceeding inter farm variability (van Beek et al., 2003) it could equally be argued that farm scale surpluses do not adequately capture the heterogeneous distribution of surpluses across farms. Longer term evaluations and / or more extensive coverage are required if field surpluses are to be more informative with regard farm nutrient management and to decipher whether hotspots stem from poor nutrient management not inherently higher surplus crops. Projected surpluses although providing greater coverage did not address these limitations.

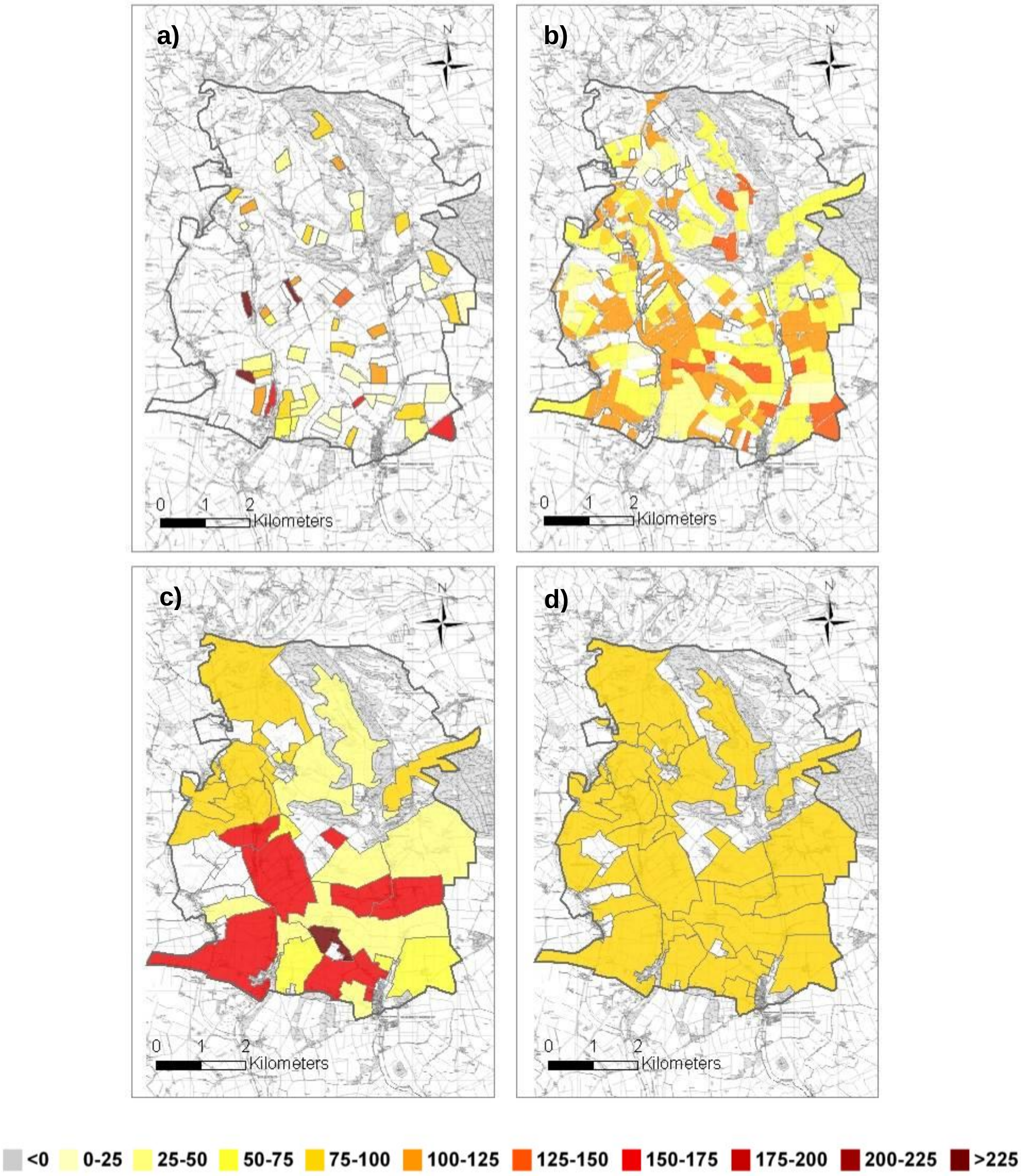


Figure 8-1: Comparison of a) soil surface b) projected soil surface c) farmgate d) catchment farmgate surpluses (kg N ha^{-1}) in MSA in 2008.

A positive relationship between organic N balances and the difference between soil surface and farmgate surpluses highlighted the absence of livestock transfer in the field scale approach (Figure 8-2); grazed grass is included as an output in the soil surface budget but only farmgate budgets fully account for the efficiency with which this and other feed stuffs are converted into livestock / livestock products. Larger farmgate (than soil surface) surpluses stemming from inefficient livestock transfers brought into question the location of surpluses connected with off field activities. (In the short term there are likely to be timelags between the production and distribution of manures produced over winter). Whole farm budgets which differentiate between farm components (field / housing) would provide more detailed assessments, allowing high risk activities to be identified and targeted, and facilitating closer links between surpluses and loss. However the benefits of capturing spatial variability must be balanced against increased timescales and computation especially when undertaking catchment scale evaluations as demanded by the WFD. Furthermore it is unlikely that budgeting would ever be conducted at a scale small enough to capture manure heaps and incidental losses which can result in significant losses.

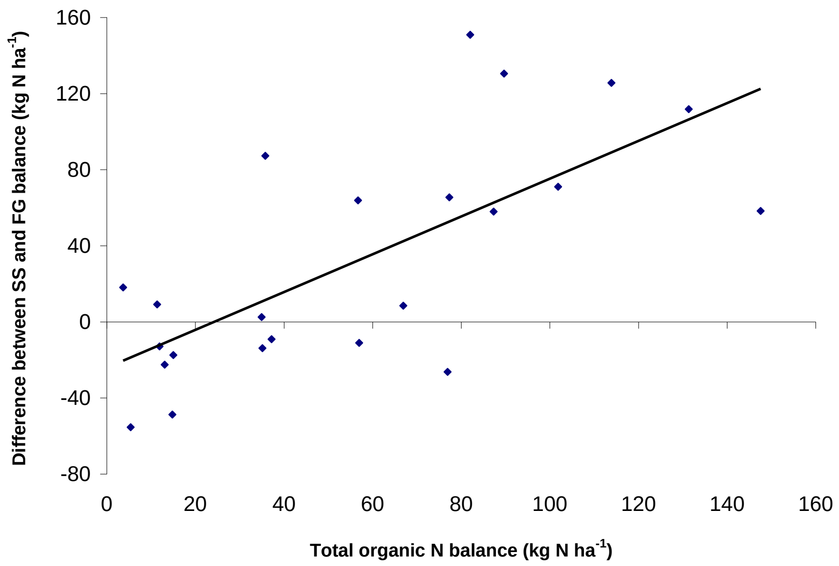


Figure 8-2: Relationship between farm manure N (kg N ha^{-1}) (imported manure – exported manure + excreta N) and the difference between the farmgate and soil surface surplus (kg N ha^{-1}). Y=0.993x-24.10 r^2=0.487 (significant to $p<0.001$).

Farm specific investigations of soil surface and farmgate budgets highlighted the significance of methodological differences namely the absence of ammonia loss in farmgate surpluses and differences in system boundaries with regard to the inclusion / exclusion of livestock transfers (discussed above). In spite of these differences, mitigation responses were largely consistent between the two approaches with soil surface and farmgate surpluses responses corresponding on 83% of farms. However, differences in the magnitude of improvements between methods highlighted differences in the sensitivity of different methodologies to different mitigation methods (Table 8-4). Soil surface (field scale) budgets were most sensitive to mitigation affecting field activities, for example measures affecting fertiliser applications, whilst activities affecting farm scale activities, for example those addressing feed inputs, were better reflected in farm scale methodologies in the short term. Accordingly changes in nutrient use observed in EMEL which mainly transpired in fertiliser application rates were better reflected in soil surface budgets. In MSA, mitigation (or other factors affecting nutrient use) affected nutrient use at both the field and farm scale resulting in similar magnitudes of change in both farmgate and soil surface budgets. Where interactions between on and off field nutrient use were greater (in MSA), soil surface and farmgate results were more alike. Choice of methodology should depend on the type / scale of mitigation being implemented and the level of interaction between field and herd (where applicable). Where mitigation is likely to affect nutrient use at both field and farm scale it is important that timelags are considered. For example changes in feed management affect the production and potential application of manure, but changes in manure application at the field scale are likely to lag behind changes in feed import at the farm scale.

Table 8-4: Effect of budget methodology on farm scale improvements. Average difference between 'before' and mitigation surpluses at the farm scale (kg N ha^{-1}). Soil surface surpluses averaged by farm.

Catchment	Soil surface surplus kg N ha^{-1}	Farmgate surplus kg N ha^{-1}
MSA	16.5	15.6
EMEL	44.2	22.0

8.3 Meeting the requirements of an assessment method

8.3.1 Evidence of a reduction in nutrient loss or nitrate concentration

With the exception of PP based evaluations, all assessment methods displayed some improvement post mitigation, however when interpreted in the context of annual variability and relative to the degree of mitigation induced change, the likelihood that change was mitigation driven was generally low. The contradictory results obtained in field scale measurement suggested an overriding sensitivity to environmental factors whilst at the catchment scale apparent improvements corresponded with differences in the distribution and total rainfall. It was also unlikely that any responses to mitigation would transpire in groundwater quality within one year. Field scale budgets exposed improvements in surpluses in EMEL and reductions in fertiliser use in MSA. However field budgets are inherently sensitive to changes in nutrient inputs and outputs occurring at the field scale. The absence of a more consistent response, and continuing over supply of N post mitigation relative to fertiliser recommendations suggest WAgriCo mitigation had little overall impact on nutrient management. In addition links could be made between favourable growing conditions and yields, and in some instances reductions in surpluses stemmed from increased output confirming sensitivity to environmental conditions. The benefits of individual mitigation methods were however apparent in field measurement and budgets, with results confirming the effectiveness of cover crops and manure management plans.

At the farm scale improvements in surpluses and efficiency were observed on a large proportion of farms, primarily stemming from a reduction in fertiliser use. Whilst the extent of improvements is encouraging, assessments were confounded by sensitivity to the price of fertiliser and feed, with reductions in fertiliser use locally corresponding with a decrease nationwide, and instances of advanced purchasing of feed. Farm scale budgets and efficiency were also influenced by changes in cropping and stocking between years. Once considered, the majority of improvements on a farm type basis were no longer thought to be a direct result of WAgriCo mitigation. However calculation of hybrid surpluses proved an objective way to account for changes in cropping, whilst changes in stocking are effectively accounted for where surpluses are presented on a on a per LU basis in addition to the usual per ha basis. On the flip side, livestock management and reductions in stocking rates represent a mitigation method in themselves. Sensitivity to changes in stocking confirmed that farmgate budgets are sensitive to a wider range of mitigation than those supported by WAgriCo. Comparison of surpluses and efficiency results highlighted a lack of sensitivity to changes in production intensity in surpluses based approaches. Where an increase in production and the implementation of mitigation coincide, efficiency based evaluations proved more useful. Differentiation between manure and fertiliser efficiency allowed evaluation of mitigation effectiveness independent on structural change (i.e. a shift in reliance between fertiliser and manure N). This is of particular relevance should fertiliser prices remain high and mitigation promote and exploit the fertiliser value of manure.

In terms of isolating responses to mitigation, the simplicity of budgets allows factors affecting budgets to be traced and quantified. Disentangling the impact of environmental conditions from measured losses is far more complex given the many processes affecting loss and the spatial variability of conditions affecting these processes. And once environmental factors have been accounted for, measurements are also affected by the factors affecting the balance of inputs and outputs at the field and farm scale. Budgets provide more opportunity to explore the causes of changes on a quantitative basis (i.e. how inputs and outputs led to a change in the surplus or efficiency), however only through talking to farmers directly can the reasons for changes and hence the extent of mitigation induced change, be fully understood. And where yield fluctuates significantly, changes in field scale inputs should be considered independent of output.

8.3.2 Sensitivity to a wide range of mitigation methods and provision of assessments at a relevant level of detail

Different assessment methods exposed sensitivity to different types of mitigation and provided different levels of assessment detail (Table 8-3). However to some extent these findings can be attributed to the scale applicability of mitigation and assessment methods. Although the implications of scale are considered in more detail in section 8.3.3, it is useful to note the limitations of assessment method sensitivity which arise as a result of the scale applicability of mitigation and assessment. Low coverage at the farm scale meant field scale budgets and measurement were most sensitive to the implementation of cover crops. In contrast the farm scale classification of mitigation codes meant their evaluation was best suited to farm scale budget and efficiency approaches, albeit no significant effect of higher mitigation codes were observed. Evaluation of mitigation code at the field scale was less applicable because EGAP classification at the farm scale did not necessarily mean higher level mitigation on all fields. Although implemented at the farm scale, responses to manure management plans were more evident in field scale budget evaluations highlighting the crop specificity of MMP impact.

Of the assessment methods investigated, all were capable of evaluating the cumulative impact of mitigation through direct comparison of pre- and post-mitigation results. Assessments of individual mitigation methods were limited to assessment methods where with vs. without comparisons were possible (i.e. field and farm scale budgets and measurement). Individual methods could not be evaluated at the catchment scale in measurement or budgets however catchment scale assessments were useful in capturing mitigation uptake. With vs. without analyses allowed evaluations to be conducted with just one year's data, thereby avoiding issues of annual variability. However evaluations of individual mitigation methods were confounded by responses to other mitigation methods. Comparisons of soil surface surpluses on spring cropped fields with / without crops suggested a positive response, however further investigation pointed to more substantial responses to fertiliser recommendations and manure management plans on farms adopting higher levels of mitigation i.e. those agreeing to grow cover crop. Separate calculation of fertiliser and manure efficiency did however allow some differentiation between responses to mitigation targeting fertiliser and manure respectively.

Differences in the mechanism underpinning methods (e.g. availability of N at source, transport of N) meant nutrient budgets and measurement were expected to differ in their sensitivity to specific mitigation methods (Cherry et al., 2008). Nutrient budgets effectively capture changes in inputs and outputs but not the assimilation and retention of nutrients whilst measurements are sensitive to changes in N availability and mitigation affecting the delivery of nutrients. Accordingly field scale surpluses failed to expose a significant response to cover crops. Only where cover crops were utilised through grazing or cutting did budgets reflect the assimilation of available N in plant material. Despite expectations that budgets were also insensitive to timing based mitigation, the benefits of delayed manure applications did transpire in surpluses.

Comparison of soil surface and farmgate surpluses at the farm scale exposed differences in the sensitivity of different budget methodologies to mitigation. Soil surface budgets were found to be more sensitive to field scale mitigation (mitigation addressing manure and fertiliser inputs) than farmgate budgets. Differences in sensitivity to manure and fertiliser mitigation were especially evident where nutrient use occurred predominately off field. Although WAgriCo mitigation did not address feed and livestock management directly, changes in feed inputs and differences in own farm manure use between farms indicated that farmgate balances are likely to be more sensitive to mitigation affecting livestock / off field activities than soil surface budgets. Farmgate surpluses were also found to better reflect the overall impact of mitigation on farms where interactions between herd and field were high.

With respect to the requirements that waterbodies reach good status, the provision of cumulative evaluations is of particular relevance; waterbodies reflect activities throughout catchment-wide and thus differentiation between responses is not required. However the iterative process of implementation, evaluation and refinement associated with PofMs requires that the most effective mitigation methods be identified to ensure intervention is cost effective. As a result field and farm scale budget and measurement capable of evaluating mitigation on an individual basis are also required. Some mitigation methods are more policy relevant than others; for example Action Programmes associated with the Nitrate Directive target fertiliser and manure inputs. Evaluations in MSA and EMEL confirmed that both measurement and budget approaches are sensitive to mitigation addressing the availability of N at source and thus to Action Programme measures. Catchment Sensitive Farming promotes a wider array of mitigation including those addressing the delivery of nutrients to waterbodies. Investigations in MSA and EMEL confirmed that budgets are less applicable where mitigation promotes nutrient retention and nutrient delivery is reduced. In general mitigation is increasingly targeting manure management making the availability of assessment methods sensitive to manure based mitigation of particularly relevance. Investigations highlighted field scale budgets as best suited to the evaluation of manure management plans. However should poor manure management be widespread, farm scale assessment methods would be increasingly sensitive to the implementation of manure related mitigation.

8.3.3 Assessment at a scale of relevance

Investigation confirmed that budget and measurement based assessments could be performed at field, farm and catchment scales, however a lack of correspondence between scales highlighted the scale specificity of processes and factors affecting results. For example PP leachate concentrations where 10 fold higher than groundwater N concentrations, a result of high N field leachate mixing with low N waters. Similarly field and farm surpluses showed little spatial correspondence with crop type explaining the majority of variation in soil surface surpluses compared to farm type which accounted for the majority of variation in farmgate surpluses. The scale of assessment also affected the certainty with which improvements could be attributed to mitigation and the likelihood of timelags (see section for more details regarding timescales). With increasing scale the range of confounding variables increases. Catchment scale measurement for example must be interpreted relative to variation in farm practice, rainfall, changes in transformations (which in turn are affected by temperature, moisture etc.) dilution, inputs from catchment scale sources etc.

Scale affected relationships between surpluses and loss, and thus the value of surpluses as an indicator of loss. Although relationships between the two approaches were generally weak, results displayed greater agreement at the field scale where losses are subject to fewer transformations, delays and opportunities for dilution or supplementation. Relationships were absent where surpluses and loss were compared across scales with farmgate surpluses and catchment scale measurement exposing little agreement. In addition to the inherent limitation of the surplus – loss relationship, larger ‘areas of influence’ associated with measurement sites meant it was not possible to relate groundwater N concentration to the management of an individual farm. Correspondence between surpluses and loss were more evident where results were presented as catchment averages. Links between measurement sites and farms are removed and the heterogeneity of physical properties evened out.

A number of scale specific advantages and applications were identified. Field scale assessments would be useful where the geographical extent of mitigation is limited and thus would be well suited to early stage assessments providing initial confirmation of mitigation potential. Detailed evaluations and targeted mitigation opportunities stemming from field scale assessment also fits well within the process of PofMs refinement and the need for cost effective action. Farm scale methods proved useful in ensuring mitigation was compatible with existing farm systems as well as highlighting inefficient systems. However the WFD is driven by a need to improve waterbody status which is determined by land management and uptake of mitigation catchment wide. PofMs are devised to reflect the agricultural and environmental conditions of specific river basins demanding catchment scale assessment methods which capture the diversity of these conditions.

8.3.4 Assessments over an appropriate timescale

Budget and measurement results reflected nutrient management beyond the year of measurement / calculation. For example previous cropping affects SMN and advance purchasing of feed and fertiliser affects farmgate budgets. However in the absence of longer term data, the implications of this could not be explicitly accounted. Short timescales also reduced the likelihood of observing responses in groundwater. Although long term groundwater monitoring exposed short term variation in N concentrations indicative of a rapid response to rainfall and potentially to changes in land management, groundwater modelling in the MSA and EMEL catchment suggests improvements would not have been observed in just one year. Indeed the effects of a zero leaching scenario were not evident in simulated MSA groundwater N concentrations until 2025 (Ruskin et al., 2008). Crucially these timescales exceed those associated with the WFD in which PofMs must be operational by 2012 and good status achieved by 2015. Coupled with 6 yearly reporting of River Basin Management Plans and associated evaluation and refinement of PofMs, the WFD requires more responsive methods of assessment.

It is also important to consider the timescales of mitigation implementation and impact. Some mitigation methods take time to be implemented whilst others impact on nitrate loss beyond the year of implementation. Changes in manure management for example will affect loss in the following year whilst long term growth of cover crops can increased the risk of loss in the long term due to accumulating organic N. Measurement based assessment highlighted the importance of mitigation being fully implemented prior to evaluation. SMN sampling for example was conducted before fertiliser applications were made and thus results did not reflect their performance. Changes in nutrient use may be evident in farm scale methods before transpiring in field scale methods, for example imports of feed or fertiliser. Assessment must respect variability and timelags in mitigation impact as well as in the evaluations which follow. Longer timescales (than those in this investigation) would increase confidence in the full impact of mitigation being captured in mitigation evaluations.

8.3.5 Respect data and resource availability

8.3.5.1 Cost and time

Economic analysis and cost effectiveness underpins the WFD and as such it is important that mitigation and assessment methods are cost effective. Although mitigation cost effectiveness was beyond the scope of this project and has been reported elsewhere e.g. Haygarth et al., (2005), Table 8-5 demonstrates the likely costs associated with each assessment method applied to MSA. On an annual basis measurement approaches are generally more expensive, however the difference increases considerably where timescales are also considered. Sensitivity to environmental condition means measurement approaches demands longer term assessment, especially at the catchment scale. In contrast, budgets can be calculated retrospectively enabling farm data from a number of years to be collected during one visit. The costs shown in Table 8-5, which assume one farm visit per year, could potentially be reduced. The collection of historic data is made more straightforward where farmers have computerised farm records. Regulatory record keeping requirements is likely to increase the efficiency of data management and transfer. However farmers must be willing to offer farm records, requiring the development of a good relationships between farmers and 'assessors', and sufficient farmer interest to supply the required information. WAgriCo demonstrated that incentives are the most effective way to generate the necessary level of interest.

Table 8-5: Comparison of costs associated with measurement and budget assessments in MSA. See Table A7 in the appendix for full calculation details and assumptions. [a] Cost would be lower where data is collected for all years at the same time.

Method	Est. annual cost in MSA	Timescales	Total cost	Specialist equipment / requirements	By who??
SMN	£2064	Medium term – 5 years	£10320	Hydrocare Vehicle with tow bar	Farmers / contractors / researchers
PP	£6930	Medium term – 5 years	£32090	Hydrocare Vehicle with tow bar	Contractors / researchers
WQ	£4440	Long term 10 years +	£44400	Pump for BH samples	Environment agency / contractors
Soil surface budget	£2880	Medium term – 5 years	£14400[a]	Good relationship with farmer	Farmers, scientists
Farmgate budget	£1440	Medium term – 5 years	£7200[a]	Good relationship with farmer	Farmers Catchment advisers
Efficiency	£2880	Medium term – 5 years	£14400[a]	Good relationship with farmer	Farmers Catchment advisers

With respect to the accountability of assessment activities, increasing emphasis on farmer engagement and catchment management has increased the number of on the ground catchment advisors, funded by CSF, Wildlife Trusts, Rivers Trusts and water companies to name but a few. Providing advice and support to farmers, and therefore building trusted relationships, catchment advisors would be well placed to obtain farm data, however they may not have the time to calculate budgets themselves, especially field scale budgets which are particularly data intense. Farmgate budget software is however being made available to farmers for example via the latest release of PLANET, the electronic version of RB209 fertiliser recommendations. If the economic benefits of nutrient budgets and efficiency are effectively communicated farmers might consider calculating budgets themselves.

Surveillance and operational monitoring requirements under the WFD, mean the Environment Agency are likely to continue long term but low spatial resolution monitoring through England and Wales. This would supply useful information regarding longer term change to support indications of change provided by nutrient budget methods. Although SMN has direct benefits for farmers by facilitating more accurate fertiliser recommendations, the expense and difficulty in obtaining representative samples may prevent voluntary monitoring. It is likely that SMN and PP monitoring would only occur where research needs demanded assurance of observed mitigation change in the medium term, and as such would be carried out by consultants / researchers and their contractors.

8.3.5.2 Synergies and secondary benefits

The need for cost effective action makes secondary benefits and synergies an important consideration. Owing to their communicability and direct link to farm management, nutrient budgets have the potential to fulfil an educational role. Nutrient budgets have been found to increase awareness, change attitudes, guide and inform future nutrient management, and encourage best practices (Goodlass et al., 2003; Halberg et al., 2005). Efficiency indicators allow environmental and economic targets to converge, highlighting the financial detriment of poor nutrient management. The relationships observed between farmgate surpluses and efficiency for example highlighted the potential for improvements in nutrient management within the inherent (in)efficiencies of a particular farm system. Crucially they also demonstrated associated financial gains - higher efficiency and lower surpluses which translate to lower input costs / higher output. With economics a major driver of change, a combined surplus and efficiency approach may prove effective for farmer engagement and to encourage change.

Budgets and efficiency approaches are increasingly representing a management tool in themselves, facilitating target driven mitigation and affording farmers greater control over the mitigation they adopt. Surplus based approaches have been adopted in the Netherlands (Ondersteijn et al., 2002; Hanegraaf and den Boer, 2003) and New Zealand (Shepherd et al., 2009) whilst efficiency represented a mitigation method in MSA / EMEL and on German dairy farms (Van Wepern, 2009). The value of efficiency results presented in chapter 7 was therefore two fold. On the one hand results were used to assess the impact of mitigation, and on the other, improvements in efficiency were rewarded to encourage and track improvements in nutrient utilisation. In terms of measurement, SMN improves the accuracy of fertiliser recommendations, allowing mitigation to be refined.

8.3.6 Practical and suited to end users

Budget and measurement approaches differed widely with respect to communicability and would therefore benefit different end users. In agreement with studies by Halberg et al., (2005) and Goodlass et al., (2003), farmers in MSA and EMEL showed a genuine interest in nutrient budgets and were keen to know how their farm performed against others in the catchment. After a little explanation most understood the nutrient budget concept and how their actions affected nutrient surpluses. This is encouraging given the comments of Halberg et al., (2005) that understanding the problem increases the likelihood that is will be addressed. However many farmers initially perceived a reduction in surpluses to automatically mean lower inputs with detrimental effects on yield, which was not well received. Measurement results received less interest and appeared too detached from field / farm management for farmers to be as interested. However measurement results are likely to be of more use to those interested in ecological responses where N concentration, and not N surpluses, are of greater relevance. Used in appropriate circumstances both surpluses and measurement would facilitate participation and co-operation as promoted by the WFD.

8.3.6.1 Control and enforcement

Budget based assessments exposed direct links between surpluses / efficiencies and individual farm management. As a result farmers could not avoid accepting ownership of high surplus and high risk areas. In contrast the sensitivity of measurement to environmental factors meant high loss / concentrations could not be directly attributed to farm management. Catchment scale measurements in particular were too detached from field management for farmers to accept any responsibility. (Farmers instead considered a history of dairy farming in the area and the ploughing out of grass following the second world war as reasons for the high nitrate concentrations observed locally.) The likelihood of nitrate loss being addressed is much greater where farmers accept responsibility for the problem, but it is also important that their efforts to improve the situation do not appear to be in vain. The advantage of budget based approaches is therefore two fold; linking management and results, but doing so in the short term. Where farmers are less willing to co-operate voluntarily, links between farm management and environmental state are necessary from a control and enforcement perspective. Opportunities to enforce restrictions on N use are limited where results cannot be linked to management on any one farm (Schroder et al., 2004).

However the flip side of this argument is that to comply with legislative targets waterbodies must reach good ecological status and not exceed the 11.3mg N l^{-1} standard. In the absence of a direct relationship between surpluses and loss and given differences in natural attenuation processes between catchments, surpluses do not translate to specific concentrations. In contrast measurements allow direct comparison between mitigation impact and legislative requirements and respect the one out all out approach adopted by the WFD. Spatial and temporal variability in groundwater nitrate concentration in MSA and EMEL highlighted the extent to which localised loss process and dilution determine observed concentration and thus the short comings of field or surplus based assessments. With respect to good ecological status, although chemical and ecological standards are not directly related, N concentration is more closely linked to ecological status than surpluses.

8.3.7 Sensitivity to agricultural and environmental conditions

Programmes of Measures respect agricultural and environmental conditions specific to each river basin. As a result assessment methods must be sensitivity to agricultural and environmental diversity. But while budgets are sensitive to agricultural condition and measurement to environmental conditions, neither is well suited to capturing both. Van der Werf and Petit (2002) advocate the use of indicators based on farm practice not environmental effect, while others (e.g. Zalidis et al., 2004) support indicators that integrate physical, chemical and biological processes. Perhaps both are valid depending on the specific role of the assessment method. Where results are to be disseminated to farmers and used to refine PofMs agricultural sensitivity should be the priority, but where assessments are required to track progress to good status, environmental sensitivity is more important.

8.3.8 Uncertainty

Uncertainty affected all the assessment methods investigated but to differing extents and from different sources. Nutrient budgets represent a simplification of complex farm systems with budget based assessments inherently uncertain to some extent. The absence of standard field or farm budgets limited confidence in the methodologies adopted, however the lack of budget based evaluations of mitigation effectiveness meant a standard methodology developed specifically for this purpose was unlikely to exist. The use of farmgate and soil surface methodologies aimed to avoid uncertainties connected with the feed – manure transfer, however in doing so the impact of mitigation was less well understood and sensitivity to some mitigation methods reduced. By differentiating between fertiliser and manure efficiency, the efficiency approach offered a more detailed picture of mitigation impact but exposed considerable structural uncertainty and a need for further refinement. Limitations surrounding the assumption of steady state were exposed by poor relationships between surpluses and loss. More complex budget methodologies which afford greater consideration to internal soil and loss processes are likely to improve estimations of loss, however this must be balanced against increased uncertainty (Watson and Atkinson, 1999).

Inconsistent and incomplete data introduced further uncertainty into budget based assessments. Due to NVZ reporting requirements fertiliser and cropping records were generally more complete than manure applications, supporting the use of farmgate surpluses which do not explicitly account for manure use. Sensitivity analysis highlighted the sensitivity of soil surface budgets to fertiliser and manure inputs (manure reliant fields only) and to yield. With missing manure data and estimated yields for some farms in 2008, uncertainty coincided with sensitivity in soil surface surpluses.

Difficulty obtaining representative samples of manure places uncertainty in the N contents adopted in budget methodologies, indeed manure samples obtained on MSA and EMEL farms exposed considerable variability in N contents. N contents of feed and fertiliser in contrast are less variable and thus more reliable. With manure imports and exports considerably smaller than the sum of field manure applications farmgate surpluses are therefore considered more reliable than soil surface budgets. Ad-hoc grain N sampling also exposed variability in crop N introducing uncertainty into both farmgate and soil surface surpluses. Sensitivity analysis exposed sensitivity to both crop and manure N which increases the significance of variability in these parameters. While the significance of manure related uncertainty was limited to manure dependant crops, the cumulative impact of parameter and input data uncertainty may exceed sensitivity to modest mitigation.

In terms of measurement based assessments, protocols and standard operating procedures ensured results were consistent and comparable to those obtained elsewhere. However logistical difficulties and error meant protocols were not always adhered to. Soil samples for example had to be couriered to laboratories for analysis at ambient temperature promoting mineralisation of organic matter. In addition a number of early season PP samples were lost, potentially underestimating leached losses. Moreover, measurements will always be site specific reflecting conditions in the immediate proximity and at the exact time of sampling. Where assessments are conducted over longer time scales it is important that 'measurement' is consistent between years. Wahlin and Grimvall (2008) suggested that long term measurement trends are more extensively influenced by changes in sampling and laboratory procedure than actual changes in the state of the environment.

While budgets represent a simplified farm system, measurements capture the complete field / catchment system response. As a result greater ambiguity and uncertainty is attached to the interpretation of measurements than that of budgets. Although parameter and input data uncertainties affect budgets, these areas are likely to receive attention and be refined. Obtaining improved values of manure N or crop N content is more straightforward than reducing measurement uncertainty or deciphering the reason for changes in water quality. And with increasing computerisation of farm records input data errors will be minimised. Ignoring the uncertainties associated with the surplus – loss relationship, budgets represent a more certain approach and one which is likely to become more reliable. However the same might not be true where the budget evaluations are required to estimate actual loss.

8.4 Recommendations

Commitment to the WFD means N loss must be reduced, regardless of how challenging this might be. While we now have a good understanding of how to reduce N loss, the difficulty lies in making this a reality and demonstrating that the necessary improvements are being made. The results presented in this study confirmed that under voluntary, self implemented mitigation, changes in nutrient management are likely to be modest. Indeed observed changes in nutrient use were largely attributed to other factors such changes in cropping and fertiliser price. Although WAgriCo mitigation was therefore unlikely to yield measurable improvement in the water quality even in the longer term, useful lessons have been learnt which can be used to inform mitigation efforts and their assessment in the future.

8.4.1 Assessment methods

Based on the findings presented in this thesis and the experiences obtained through involvement in WAgriCo the following conclusions and recommendations are made regarding use of the assessment methods for short term evaluations of mitigation effectiveness:

- Field scale measurements were highly sensitive to changes in environmental conditions better suiting them to longer term assessments where annual variability evens out. Only where mitigation can be evaluated on a 'with vs. without' basis using data from a single year can the approach be used in the short term.

- Providing integrated responses to mitigation across the catchment, catchment measurements demand relatively low spatial resolution sampling at low cost however evaluations must be performed over the longer term resulting in high total cost. With timescales extending beyond those demanded by the WFD, it is suggested that long term measurement of catchment waterbodies be adopted as a secondary measure to provide confirmation that indirect improvements (i.e. reduced surpluses) translate to improvements in water quality in the longer term. With measurements providing assessments on an N concentration basis catchment scale measurement is preferable to catchment scale budgets, and with similar monitoring routinely undertaken by the EA, additional sampling might be avoidable.

- Field scale budgets captured change in inputs and outputs at relatively low cost, providing an indication of the level of mitigation induced change, highlighting over supply and confirming the effectiveness of manure management plans. However the interpretation of surpluses was confounded by annual variability in yield and cropping (when shown spatially). In the absence of longer term data individual inputs must be evaluated to ensure improvements / deteriorations are not concealed by changes in output. However given the absence of field – livestock transfers in field budgets farm scale budgets / efficiency are preferable to ensure complete mitigation responses and associated feedbacks are captured.

- Calculated at a scale of relevance to both farmers and researchers, farmgate budgets and efficiency effectively captured whole system responses and feedbacks, accounting for crop rotations and strategic crop interactions. With high levels of farmer accountability and clear links to economic drivers, farmgate surpluses and efficiency aid farmer engagement which is key to reducing N loss. Coupled with low cost and increasing opportunities for farmers calculate budgets themselves via computer programmes such as PLANET, the use of farmgate budgets and efficiency is advocated where data availability allows. Although sensitive to changes in cropping, stocking and input prices, much of this can be accounted for where results are interpreted relative to economic factors. Although longer term assessments would increase the certainty with which improvements could be attributed to mitigation, in contrast to measurement, budgets provide a more rapid response to changes in nutrient use which is important given the short timescales under which the WFD is being implemented. It would however be necessary to support budget / efficiency calculations with long term measurement due to the indirect relationship between surpluses and loss.

- It is also proposed that farmgate surpluses and efficiency are used together to exploit opportunities to reduce surpluses through improved efficiency, and demonstrate associated financial gains which are likely to engage farmers and initiate change. A combined approach would also respect changes in production intensity; efficiency does not penalise farms increasing their production intensity at the same time as implementing mitigation. Where structural change occurs (e.g. increased reliance on manure N) use efficiency is preferable where source specific nutrient efficiency calculations overcome the inherent differences in nutrient efficiency attached to fertiliser and manure.

It is therefore suggested that a combined approach is adopted. Farmgate surpluses and efficiency should represent the primary assessment methods

to evaluate cumulative mitigation impact, supported by longer term water quality measurements. Evaluation of field scale mitigation is best achieved using field scale measurement or budgets on a with vs. without basis which avoid annual variability.

8.4.2 Mitigation methods

Although the range of WAgriCo mitigation methods was limited, their implementation highlighted a number of issues relevant to future mitigation.

Fertiliser applications were already lower than recommended levels where only inorganic N was applied generating little opportunity for further reductions without compromising yield. With fertiliser applications higher in EMEL and MSA than nationally there is even less opportunity for reductions elsewhere. However where manure was applied (on 88% MSA and EMEL farms and 69% farms nationally) nutrient inputs were supra-optimal exposing considerable opportunities to reduce fertiliser inputs through improved manure accounting. Although in most cases inputs remained above recommended level post mitigation, the dissemination of fertiliser recommendations highlighted the extent of oversupply. Where both fertiliser and manure is applied and the likelihood of supra-optimal inputs high, fertiliser recommendations have the potential to inform and improve nutrient management. However the failure of WAgriCo mitigation to tackle over supply in MSA and EMEL confirmed that some farmers did not accept the fertiliser value of manure, and that despite being supplied with tailored recommendations, some farmers chose to disregard them. This was despite NVZ designation which requires fertiliser inputs, soil supply and crop demand to be balanced. Associated cost savings should be used to promote complete accounting of manure N in fertiliser planning, and more effort made to understand why recommendations were not followed. (WAgriCo tried to achieve this but few responses were received). Expansion of NVZs would be expected to increase compliance with fertiliser recommendations and reduce instances of oversupply assuming requirements are adhered to.

Improved manure management (MMPs, reduced manure inputs, delayed manure inputs and complete accounting of manure N) was consistently identified as being effective in improving nutrient management at the field, farm and catchment scale. This is consistent with wider effort to promote the value of manure and improve its management for example through CSF, NVZ Action Programmes and MANNER.

Results also highlighted targeted responses to MMPs, addressing high risk fields e.g. those in maize. However farm data confirmed that excessively large manure applications were still being made on some MMP farms highlighting a need for greater guidance to ensure mitigation is fully implemented. Longer and more widely applicable closed periods in NVZs may exacerbate this problem; catchment advisors should try target these farms and emphasis the value of manure and the impacts of oversupply. Where farms do not respond to a supportive approach, enforcement or penalties would be required.

In agreement with a weight of previous evidence, investigations confirmed the effectiveness of cover crops in reducing N loss. However similar to MMP implementation, farm data exposed opportunities for improved establishment and management and thus a need for greater advice and support. While their applicability is limited, cover crops covered less than half the spring cropped area exposing considerable opportunities to increase uptake. However farmers voiced some negativity around their establishment and detrimental effects on the following spring crop.

Analysis of farm data highlighted that MSA and EMEL farms would have benefitted from mitigation addressing feed management. On a number of farms reductions in fertiliser were counteracted by increased feed. Balancing feed N with livestock requirements and increasing reliance on maize have been noted as effective mitigation options (Cuttle et al., 2006). However in contrast to mitigation targeting manure and fertiliser, uptake would be limited in predominately arable areas.

Across all methods, greater impact may have been observed where financial back up existed. In the absence of potential compensation farmers were reluctant to take too greater risks. In general farmers favoured simple, low cost, look good mitigation, the implementation of which was supported by relevant, sound advice. In the presence of considerable existing legislation and bureaucratic requirements, farmers were keen to avoid additional regulation. However where mitigation remains voluntary those behaving the worst are unlikely to be those most involved.

8.4.3 Farmer engagement

Farmer buy-in within WAgriCo was relatively high with approximately two thirds of EMEL and MSA (by area) participating in the WAgriCo project. However this figure and the level of involvement would almost certainly have been lower had financial incentives not been offered; a large proportion of farmers implemented mitigation primarily for money received in return. Incentives are therefore key to active farmer involvement, however these do not necessarily have to be financial. Payments in kind were also found to be effective, for example farmers received the results of field and catchment monitoring in return for allowing the samples to be taken. WAgriCo farmers were also keen to obtain a practical and economic advantage, recognising more efficient management would save them money and potentially increase profits above their peers.

Although primarily financially motivated, WAgriCo farmers were genuinely interested in the state of their catchment, and keen to see the results of monitoring activities. However this did not mean they were willing to accept responsibility for less favourable results, with many farmers attributing high N concentrations in groundwater to a history of dairy farming and the ploughing up of grassland post World War 2. Field and farm scale results were therefore useful in that they retained a degree accountability. Indeed farmers were very interested in the their farm surpluses and keen to know how they compared to their peers. Farmers were also keen to see proof, proof that they were the cause (or not) of high N concentrations, and proof that mitigation brought positive change. Understandably they did not want to make changes for no reason especially where it might be costly or inconvenient. Budgets and efficiency results are, therefore, of particular relevance, linked directly to farms and responsive to changes in nutrient use in the short term.

Although a few farmers were very proactive and keen to pursue new technologies and conservation techniques, most were initially wary of WAgriCo. The level of participation benefitted from the gradual building of trusted relationships with knowledgeable catchment advisors. However there were some issues of apparently conflicting messages between from WAgriCo and CSF catchment advisors. To instil key messages, retain confidence and exploit parallels continuity is important. Following on from this, farmers did not like the short term nature of catchment management initiatives. Farmer were reluctant to make significant changes without long term commitments of support. Short term projects such as WAgriCo do not exploit the good relationships built up with farmers, the information obtained and resources installed at considerable cost, for example porous pots.

8.5 Future work

Investigations into the usefulness of budget and measurement based assessment methods would be benefit from further work in a number of areas.

- Nutrient budgets were sensitive to changes in external factors such as fertiliser price and yield. Budget based evaluations would benefit from quantification of variables known to have significant implications on surpluses. In doing so changes in surpluses could be interpreted more objectively.

- Field scale nutrient surpluses exposed surplus hotspots, however when evaluated on an annual basis it was not possible to differentiate between the influence of crop type and field / farm specific nutrient management. Investigations into the usefulness of field scale budgets would benefit from calculation across complete crop rotations to investigate whether the significance of nutrient management on field surpluses can be discerned from that of crop type.

- Relationships between surpluses and loss were generally weak, however this may reflect the relatively short time scale over which results were available. Studies reported in the literature suggest longer term investigations are required to expose the links between surpluses and loss. Confidence that surplus reductions transpire in measured loss would benefit from longer term investigations into the relationship between surpluses and loss in the context of mitigation evaluation (which places greater importance on corresponding improvements than absolute losses). Where / if strong relationships are observed, opportunities to account for transport processes and estimate actual losses should be explored.

- Application of the modified efficiency methodology to MSA / EMEL farms and comparison of efficiency results with farmgate budget results highlighted a need for further methodological refinement. Increased sensitivity of fertiliser efficiency to mitigation is required to improve evaluations of manure mitigation and heighten sensitivity to fertiliser mitigation. Further work is also required to balance flows and ensure consistency across manure and fertiliser components.

- Relationships between surpluses and efficiency exposed opportunities to reduce surpluses through improved efficiency. It would be interesting to calculate likely cost savings associated with improvements in efficiency / reductions in surpluses, and quantify more explicitly the changes in management required to achieve such improvements in efficiencies.

- The investigations undertaken assessed the effectiveness of / sensitivity to a limited number of mitigation options which resulted in modest mitigation induced change. Assessments of evaluation methods would benefit from broader investigations covering a wider range of mitigation options to ensure the conclusions presented here are more widely applicable. Evaluation of more drastic mitigation would be useful to reduce uncertainty in short term assessments and to increase confidence in the conclusions presented here. Increased confidence would also be achieved where assessments were performed over the longer term especially given the single year's mitigation data.

References

Aarts, H.F.M., Habekotte, B. and van Keulen, H. (2000) Nitrogen (N) management in the 'De Marke' dairy farming system. *Nutrient Cycling in Agro-ecosystems* **56**(3): 231-240.

Acheson, E.D. (1985) Nitrate in drinking water. Letter to Regional Medical Officers from Chief Medical Officer. CMO **85** 14, HMSO, London.

ADAS (1999) Development of profitable and robust dairy systems with an acceptable balance of emissions. Year 2 report for Defra project LK0613.

ADAS (2008) PLANET nutrient management software V2. Developed for Defra project IF0141.

Addiscott, T. M. and Benjamin, N. (2004) Nitrate and human health. *Soil Use and Management* **20**: 98-104.

Alfaro, M., Salazar, F.S., Oenema, O., Iraira, S., Teuber, N., Ramirez, L. and Villarroel D. (2009) Nutrient balances in beef cattle production systems and their implications for the environment. *Journal of soil science plant nutrition* **9**(1): 40-54.

Armstrong, A.C. and Burt, T.P. (1993) Nitrate losses from agricultural land. In: Burt, T.P., Heathwaite, A.L. and Trudgill, S.T. (Eds.) *Nitrate: processes, patterns and management*. Wiley, Chichester, UK. p444.

Arnold, J.G., Srinivasan, R., Muttiah, R.S. and Williams, J.R. (1998) Large area hydrologic modelling and assessment – Part I: model development. Journal of *American Water Resources Association* **34**(1): 73-89.

Aronsson, H., Tortensson, L. and Bergstrom, L. (2007) Leaching and crop uptake of N, P and K from organic and conventional cropping systems on clay. Soil Use and Management **23**: 71-81.

Avery, A.A. (1999). Infantile methemoglobinemia: Re-examining the role of drinking water nitrates. *Environmental Health Perspectives* **107**: 583-586.

Bailey, R.J. and Spackman, E (1996) A model for estimating soil moisture changes as an aid to irrigation scheduling and crop water-use studies: I. Operational details and description. *Soil Use and Management* **12**: 122-128.

Bassanino, M., Grignani, C., Sacco, D. and Allisiardi, E. (2007) Nitrogen balances at the crop and farm-gate scale in livestock farms in Italy. *Agriculture, Ecosystems and Environment* **122**: 282-294.

Bathing Waters Directive (previously 1976/160/EEC but recently updated to 2006/7/EC) *Official Journal of the European Communities* **L31**: 1-7 amended by Official Journal of the European Communities **L64**: 37-51.

Beaudoin, N., Saad, J.K., van Laethem, C., Machet, J.M., Maucorps, J. and Mary, B (2005) Nitrate leaching in intensive agriculture in Northern France: Effects of farming practices, soil and crop rotations. *Agriculture, Ecosystems and Environment* **111**: 292-310.

Bechmann, M., Eggestad, H.O. and Vagstad, N. (1998) Nitrogen balances and leaching in four agricultural catchments in south-eastern Norway. *Environmental Pollution* **102**:493-499.

Bechmann, M.E., Berge, D., Eggestad, H.O. and Vandsemb, S.M. (2005) Phosphorus transfer from agricultural areas and its impact on the eutrophication of lakes - two long-term integrated studies from Norway. *Journal of Hydrology* **304**(1-4): 238-250.

Bechmann, M., Deelstra, J., Stalnacke, P., Eggestad, H.O., Oygarden, L. and Pengerud, A. (2008) Monitoring catchment scale agricultural pollution in Norway: policy instruments, implementation of mitigation methods and trends in nutrient and sediment loss. *Environmental Science and Policy* **II** 102-114.

Beckwith, C.P., Cooper, J., Smith, K.A. and Shepherd, M.A. (1998) Nitrate leaching loss following application of organic manures to sandy soils in arable cropping. I. Effects of application time, manure type, overwinter crop cover and nitrification inhibition. *Soil Use and Management* **14**:123–130.

Berentsen, P.B.M. and Tiessink, M. (2003) Potential effects of accumulating environmental policies on Dutch Dairy farms. *Journal of Dairy Science* **86**: 1019-1028.

Beresford, S.A. (1985) Is nitrate in drinking water associated with gastric cancer in the urban UK? *International Journal of Epidemiology* **14**: 57-63.

Berg, W.E., Brunsch, R., Eurich-Menden, B., Dohler, H., Dammgen, U., Osterburg, B. and Bergschmidt, A. (2003). Ammonia emissions from German animal husbandry. 3rd International Conference on Air Pollution from Agricultural operations. 12-15 October, 2003. Raleigh, NC, USA.

Berry, P.M., Stockdale, E.A., Sylvester-Bradley, R., Philipps, L., Smith, K.A., Lord, E.I., Watson, C.A. and Fortune, S. (2003) N, P and K budgets for crop rotations on nine organic farms in the UK. *Soil Use and Management* **19**(2): 112-118.

Bleken, M. A. (2009) Use of material flow analysis to evaluate the N use efficiency of farming systems. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.385-6.

Bleken, M.A., Steinshamn, H. and Hansen, S. (2005) High nitrogen costs of dairy production in Europe: Worsened by intensification. *Ambio* **34**(8) 598-606.

Borja, A. and Elliot, M. (2007) What does ‘good ecological potential mean, within the European Water Framework Directive? *Marine Pollution Bulletin* **54**: 1559-1564.

Borsting, C.F., Kristensen, T., Misciattelli, L., Hvelplund, T. and Weibbjerg, M.R. (2003) Reducing nitrogen surplus from dairy farms. Effects of feeding and management. *Livestock production science* **83**: 165-178.

Bowes, M.J., Smith, J.T. and Neal, C. (2009) The value of high resolution nutrient monitoring: A case study of the River Frome, Dorset, UK. *Journal of Hydrology* **378**: 82-96.

Brouwer F. (1998) Nitrogen balances at farm level as a tool to monitor effects of agri-environmental policy. Nutrient Cycling in Agro-ecosystems **52**(2-3): 303-308.

Brown, L., Scholefield, D., Jewkes, E.C., Lockyer, D.R. and del Prado, A. (2005a) NGAUGE: A decision support system to optimise N fertilisation of British grassland for economic and environmental goals. Agriculture Ecosystems and Environment **109**(1-2): 20-39.

Brown, R., Grundy, J. and Roy, S. (2005b) State of the Aquifer Report: Frome and Piddle (Dorset) Groundwater Monitoring Unit. Prepared for the Environment Agency. Available at: http://www.environment-agency.gov.uk/static/documents/Business/GW_Aquifer_Quality_Reports_180510.pdf (Accessed 27 August 2010).

BSFP (2009) The British survey of fertiliser practice 2008. Prepared for Defra. Available at: http://www.defra.gov.uk/evidence/statistics/foodfarm/enviro/fertiliserpractice/index.htm (Accessed 8 September 2010).

Burt, T.P. (2001) Integrated management of sensitive catchment systems. *Catena* **42**: 275-290.

Bussink, D.W. (1992) Ammonia volatilisation from grassland receiving nitrogen fertiliser and rotationally grazed by dairy cattle. *Fertiliser research* **33**: 257-265.

Cadee, G.C. (1990) Increase in Phaeocystis blooms in the westernmost inlet of the Wadden Sea, the Marsdiep, since 1973. In: Lancelot, C., Billen, G. and Barth, H. (Eds.) Eutrophication and Algal Blooms in North Sea coastal zones, The Baltic and Adjacent Sea Areas: Predictions and assessment of preventative actions, CEC Water Pollution Research Reports, 12, Guyot, Brussels pp. 102-112.

Campling, P., Terres, J.M., Vande Walle, S., van Orshoven, J. and Crouzet, P. (2005) Estimation of nitrogen balances from agriculture for EU-15: spatialisation of estimates to river basins using the CORINE Land Cover. *Physics and Chemistry of the Earth* **30** 25-34.

Catt, J.A., Christian, D.G., Goss, M.J., Harris, G.L. and Howse, K.R. (1992) Strategies to reduce nitrate leaching by crop rotation, minimal cultivation and straw incorporation in the Brimstone Farm experiment, Oxfordshire. *Aspects of Applied Biology (Nitrate and Farming Systems)* **30**: 255-262.

Catt, J.A., Howse, K.R., Christian, D.G., Lane, P.W., Harris, G.L. and Goss, M.J. (1998) Strategies to decrease nitrate leaching in the Brimstone Farm Experiment, Oxfordshire, UK, 1988-1993: the effects of winter cover crops and unfertilised grass leys. *Plant and Soil* **203**: 57-69.

Chambers, B.J., Lord, E.I., Nicholson, F.A. and Smith, K.A. (1999) Predicting nitrogen availability and losses following application of organic manures to arable land: MANNER. *Soil Use and Management* **15**: 137-143.

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

Cherry, K.A., Shepherd, M.A., Withers, P.J.A. and Mooney, S.J. (2008) Assessing the effectiveness of actions to mitigation nutrient loss from agriculture: A review of methods. *Science of the Total Environment* **406**: 1-23.

Collins, A.L., Stromqvist, J., Davison, P.S. and Lord, E. I. (2007) Appraisal of phosphorus and sediment transfer in three pilot areas identified for the catchment sensitive farming initiative in England: application of the prototype PSYCHIC model. *Soil Use and Management* **23**(1): 117-132.

Collins, A.L. and McGonigle, D.F. (2008) Monitoring and modelling diffuse pollution from agriculture for policy support: UK and European experience. *Environmental science and policy* **II**: 97-101.

Cornblath, M. and Hartmann, A.F. (1948) Methaemoglobinaemia in young infants. *Journal of Paediatrics* **33**: 421-425.

Croll, B.T. and Hayes, C.R. (1988) Nitrate and water supplies in the United Kingdom. *Environmental Pollution* 50(1-2): 163-187.

Cuttle, S.P. and Scholefield, D. (1995) Management options to limit nitrate leaching from grassland. *Journal of Contaminant Hydrology* **20**: 299-312.

Cuttle, S.P., Scurlock, R.V. and Davies, B.M.S. (2001) Comparison of fertiliser strategies for reducing nitrate leaching from grazed grassland with particular reference to the contribution from urine patches. *The Journal of Agricultural Science* **136**(2): 221-230.

Cuttle, S., Shepherd, M.A. and Goodlass, G. (2003) A review of leguminous fertility –building crops, with particular reference to nitrogen fixation and utilisation. Prepared for Defra project OF0316.

Cuttle, S.P., Shepherd, M.A., Lord, E.I. and Hillman, J. (2004) Literature review of the effectiveness of measures to reduce nitrate leaching from agricultural land. Prepared as part of Defra project NT2511.

Cuttle, S.P., MacLeod, C.J.A., Chadwick, D.R., Scholefield, D., Haygarth, P,M,, Newell-Price, P., Harris, D., Shepherd, M.A., Chambers, B.J. and Humphrey, R. (2006) An Inventory of Methods to Control Diffuse water Pollution from Agriculture: User Manual. Prepared as part of Defra project ES0203. Available at http://randd.defra.gov.uk/ (Accessed 8 September 2010).

Dalgaard, T., Heidmann, T., and Mogensen, L. (2002) Potentail N-losses in three scenarios for conversion to organic farming in a local area of Denmark. *European Journal of Agronomy* **16**: 207-217.

Defra (2000) Reference Book (RB) 209 Fertiliser Recommendations for Agricultural and Horticultural Crops 7th Edition. Replaced by Fertiliser Manual (RB209) 8th Edition June 2010. Available at http://www.defra.gov.uk/foodfarm/landmanage/land-soil/nutrient/documents/rb209-rev-100609.pdf (Accessed 26 August 2010).

Defra (2001a) *Water Framework Directive* (online) Available at: http://www.defra.gov.uk/environment/quality/water/wfd/ (Accessed 26 August 2010).

Defra (2001b) *Water Framework Directive: Characterisation* (online). Available at: http://www.defra.gov.uk/environment/quality/water/wfd/characterisation.htm (Accessed 26 August 2010).

Defra (2002a) *Implementing the Nitrates Directive in England* (online). Available at: http://www.defra.gov.uk/environment/quality/water/waterquality/diffuse/nitrate/directive.htm (Accessed 26 August 2010).

Defra (2002b) Nitrous oxide and denitrification measurements on the nutrient demonstration farms – Final report of Defra project CC0238. Available at http://randd.defra.gov.uk/ (Accessed 26 August 2010).

Defra (2005) Farm Nutrient Auditing: Support to PLANET (Benchmarking). Final report of Defra project ES0124. Available at http://randd.defra.gov.uk/ (Accessed 26 August 2010).

Defra (2006) Nutrient management decision support system (PLANET). Defra project KT0113. Available at http://www.planet4farmers.co.uk. (Accessed 8 September 2010).

Defra (2007) *Catchment Sensitive farming* (online) http://www.defra.gov.uk/foodfarm/landmanage/water/csf/index/htm (Accessed 26 August 2010).

Defra (2008) June Survey of Agriculture and Horticulture in England 2008. Available at: http://www.defra.gov.uk/evidence/statistics/foodfarm/landuselivestock/junesurvey/results.htm (Accessed 8 September 2010).

Defra (2009) Joint announcement by the agricultural departments of the United Kingdom. Cereal and oilseed rape production estimates: 2008 Harvest United Kingdom – Final Results. Last accessed 12/08/2009. Now replaced by 2009 Harvest results. Available at http://www.defra.gov.uk/evidence/statistics/foodfarm/food/cereals/cerealsoilseed.htm (Accessed 26 August 2010).

Defra (2010a) Nitrate Vulnerable Zones (England 2009). Available at: http://web.adas.co.uk/defra/ (Accessed 26 August 2010).

Defra (2010b) *Further information: Cross Compliance* (online). Available at http://www.defra.gov.uk/foodfarm/farmmanage/singlepay/furtherinfo/crosscomply/index.htm (Accessed 8 September 2010).

de Koeijer, T.J., Wossink, G.A.A., Smit, A.B., Janssens, S.R.M., Renkema, J.A. and Struik, P.C. (2003) Assessment of the quality of farmers' environmental management and its effects on resource use efficiency: a Dutch case study. *Agricultural Systems* **78**: 85-103.

D'Haene, K.D., and De Mey, K. (2009) Towards higher N efficiency through an integrated monitoring tool for sustainable farming. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.581-2.

Domburg, P., Edwards, A.C., Sinclair, A.H., Wright, G.G. and Ferrier, R.C. (1998) Changes in fertiliser and manorial practices during 1960- 1990: implications for N and P inputs to the Ythan catchment, N.E. Scotland. Nutrient Cycling in Agro-ecosystems **52**: 19-29.

Domburg, P., Edwards, A.C., Sinclair, A.H. and Chalmers, N. (2000) Assessing nitrogen and phosphorus efficiency at farm and catchment scale using nutrient budgets. Journal of Science, Food and Agriculture **80**(13): 1946-1952.

Donohue, I., Mcgarrigle, Mills, P. (2006) Linking catchment characteristics and water chemistry with the ecological status of Irish rivers. *Water Research* **40**: 91-98.

Drinking Water Directive (98/83/EC) standard set in 75/440/EEC, *Official Journal of the European Communities* **L194**: 26–31 (1975). Replaced by 80/778/EEC, *Official Journal of the European Communities* **L229**: 11–29 (1980) Amended by 98/83/EC, *Official Journal of the European Communities* **L330**: 32–54, (1998), but standard unchanged.

Edwards, A.C., Twist, H. and Codd, G.A. (2000) Assessing the impact of terrestrially derived phosphorus on flowing water systems. Journal of Environmental Quality **29**(1): 117-124.

Edwards, A.C. and Withers, P.J.A. (2007) Linking phosphorus sources to impacts in different types of water body. *Soil Use and Management* **23**: 133-143.

Environment Agency (2006) The State of Groundwater in England and Wales, Environment Agency, Bristol, pp 24. Available at: http://publications.environment-agency.gov.uk/epages/eapublications.storefront (Accessed 8 September 2010).

Environment Agency (2008) Indicator DA3: Nitrate and phosphorus levels in rivers. Available at: http://www.defra.gov.uk/evidence/statistics/foodfarm/enviro/observatory/indicators/d/da3_data.htm. (Accessed 26 August 2010).

Environment Agency (2009) River Basin Management Plan South West River Basin District. Available at: http://www.environment-agency.gov.uk/research/planning/33106.aspx (Accessed 26 August 2010).

Environment Agency (2010) *Nitrate Vulnerable Zones, 2002* (online). Available at: http://www.environment-agency.gov.uk/cy/ymchwil/llyfrgell/data/58801.aspx (Accessed 28 August 2010).

Eulenstein, F., Werner, A., Willms, M., Juszczak, R., Luis Schlingwein, S., Chojnicki, B.H. and Olejnik, J (2008) Model based scenario studies to optimize the regional nitrogen balance and reduce leaching of nitrate and sulphate of an agriculturally used water catchments*. Nutrient cycling in Agro-ecosystems* **82**: 33-49.

European Commission (EC) (2001) Common Implementation Strategy for the Water Framework Directive (2000/60/EC) available at http://ec.europa.eu/environment/water/water-framework/objectives/pdf/strategy.pdf (Accessed 26 August 2010).

European Commission (EC) (2010a) Report from the Commission to the council and the European Parliament On implementation of Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources based on Member State reports for the period 2004-2007 (COM(2010)47). Available at: http://ec.europa.eu/environment/water/water-nitrates/index_en.html
(Accessed 26 August 2010).

European Commission (EC) (2010b) *Introduction to the new EU Water Framework Directive* (online). Available at: http://ec.europa.eu/environment/water/water-framework/info/intro_en.htm (Accessed 26 August 2010).

European Commission (EC) (2010c) *Ecological status and intercalibration* (online). Available at: http://ec.europa.eu/environment/water/water-framework/objectives/status_en.htm (Accessed 26 August 2010)

European Commission (EC) (2010d) *Implementation of Nitrate Directive* (online). Available at: http://ec.europa.eu/environment/water/water-nitrates/index_en.html (Accessed 26 August 2010)

European Commission (EC) (2010e) *Urban waste water directive overview* (online). Available at: http://ec.europa.eu/environment/water/water-urbanwaste/index_en.html (Accessed 26 August 2010).

European Environment Agency (EEA) (2004) EEA Indicator Fact Sheet (WEU1) Nitrate in Groundwater. Available at: https://www.eea.europa.eu/data-and-maps/indicators/nitrate-in-groundwater-1 Accessed 26 August.

European Environment Agency (EEA) (2005) Source apportionment of nitrogen and phosphorus inputs into the aquatic environment. EEA Report No 7/2005. Available at http://www.eea.europa.eu/publications/eea_report_2005_7 (Accessed 8 September 2010).

European Environment Agency (EEA) (2009) *Nutrients in freshwater (CSI 020)* (online). Available at: https://www.eea.europa.eu/data-and-maps/indicators/nutrients-in-freshwater/nutrient-in-freshwater-assessment-published-1 (Accessed 26 August 2010).

European Fertiliser Management Association (2010) Forecast of food, farming and fertiliser use in the European Union 2005-2015. Available at: http://www.efma.org (Accessed 28 August 2010).

Fezzi, C., Rigby, D., Bateman, I.J., Hadley, D. and Posen, P (2008). Estimating the range of economic impacts on farms of nutrient leaching reduction policies. *Agricultural Economics* **39**: 197-205.

Fish Breeding Directive (2006/44/EC) *Official Journal of the European Communities* **L264**: 1-12.

Flynn, N.J., Paddison, T. and Whitehead, P.G. (2002) INCA Modelling of the Lee System: Strategies for the reduction of nitrogen loads. *Hydrology and Earth System Sciences* **6**: 467–483.

Forman, D., Al-Dabbagh, A. and Doll, R. (1985) Nitrate, nitrite and gastric cancer in Great Britain. *Nature* **313**: 620-625.

Giambalvo, D., Amato, G., Di Miceli, G., Frenda, A.S., Stringi, L. (2009) Nitrogen efficiency in wheat as affected by crop rotation, tillage and N fertilisation. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.251-2.

Gitau, M.W., Gburek, W.J. and Jarrett, A.R. (2005) A tool for estimating BMP effectiveness for phosphorus pollution control. *Journal of Soil and Water Conservation* **60**(1): 1-10.

Gomann, H., Kreins, P., Kunkel, R. and Wendland, F. (2005) Model based impact analysis of policy options aimed at reducing diffuse pollution by agriculture – a case study for the river Ems and a sub catchment of the Rhine. *Environmental Modelling and Software* **20**: 261-271.

Goodchild, R.G. (1998) EU Policies for the reduction of nitrogen in water: the example of the Nitrates Directive. *Environmental Pollution* **102** S1: 737-740.

Goodlass, G., Halberg, N., Verschuur, G. (2003) Input output accounting systems in the European community – an appraisal of their usefulness in raising awareness of environmental problems. *European Journal of Agronomy* **20**:17-24.

Goulding, K.W.T. (1990) Nitrogen deposition to land from the atmosphere. *Soil Use and Management* **6**: 61-63.

Goulding, K.W.T., Bailey, N.J. and Bradbury, N.J. (1998) A modelling study of nitrogen deposited to arable land from the atmosphere and its contribution to nitrate leaching. *Soil Use and Management* **14**: 70-77.

Goulding, K.W.T. (2000) Nitrate leaching from arable and horticultural land. *Soil Use and Management* **16**: 145-151.

Goulding, K.W.T., Poulton, P.R., Webster, C.P. and Howe, M.T. (2000) Nitrate leaching from the Broadbalk Wheat Experiment, Rothamstead, UK, as influenced by fertiliser and manure inputs and weather. *Soil Use and Management* **16**:244-250.

Granlund, K., Raike, A., Ekholm, P., Rankinen, K., and Rekolainen, S. (2005) Assessment of water protection targets for agricultural nutrient loading in Finland. *Journal of Hydrology* **304**(1-4): 251-260.

Grizzetti, B., Bouraoui, F., de Marsily, G. and Bidoglio, G. (2005) A statistical method for source apportionment of riverine nitrogen loads. *Journal of Hydrology* **304**(1-4): 302-315.

Grylls, J.P., Webb, J. and Dyer, C.J. (1997) Seasonal variation in response of winter cereals to nitrogen fertiliser and apparent recovery of fertiliser nitrogen on chalk soils in southern England. *Journal of Agricultural Science* **128**: 251-262

Gutierrez, A. and Baran, N. (2009) Long term transfer of diffuse pollution at catchment scale: Respective roles of soil, and the unsaturated and saturated zones (Brevilles, France) *Journal of Hydrology* **369**: 381-391.

Haas, G., Caspari, B. and Kopke, U. (2002) Nutrient cycling in organic farms: stall balance of a suckler cow herd and beef bulls. *Nutrient cycling in agro-ecosystems* **64**: 225-230.

Habitats Directive (92/43/EEC) *Official Journal of the European Communities* **L206**: 7-50.

Halberg, N. (1999) Indicators of resource use and environmental impact for use in a decision aid for Danish livestock farmers. *Agriculture, Ecosystems and Environment* **76**: 17-30.

Halberg, N., Verschuur, G. and Goodlass, G. (2005) Farm level environmental indicators; are they useful? An overview of green accounting systems for European farms. *Agriculture, Ecosystems and Environment* **105**: 195-212.

Hanegraaf, M.C. and den Boer, D.J. (2003) Perspectives and limitation of the Dutch minerals accounting system (MINAS). *European Journal of agronomy* **20**: 25-31

Hansen, E.M. and Djurhuus, J. (1997) Nitrate leaching as influenced by soil tillage and catch crop. *Soil and Tillage Research* **41**: 203-219.

Harris, G. and Heathwaite, A.L. (2005) Inadmissible evidence: knowledge and prediction in land and riverscapes. *Journal of Hydrology* **304**(1-4): 3-19.

Hatano, R., Shinano, T., Taigen, Z., Okubo, M. and Zuowei, L. (2002) Nitrogen budgets and environmental capacity in farm systems in a large-scale karst region, southern China. *Nutrient cycling in agro-ecosystems* **63**: 139-149.

Haygarth, P.M., Granger, S., Chadwick, D., Shepherd, M.A. and Fogg, P. (2005) A Provisional Inventory of Diffuse Pollution Losses. Report for Defra project NT2511. Available at: http://randd.defra.gov.uk/ (Accessed 26 August 2010).

Heathwaite, A.L. (1993) Nitrogen cycling in surface waters and lakes. In: Burt, T.P., Heathwaite, A.L. and Trudgill, S.T. (Eds.) *Nitrate: processes, patterns and management.* Wiley, Chichester, UK, p444.

Heathwaite, A.L., Quinn, P.F. and Hewett, C.J.M. (2005a) Modelling and managing critical source areas of diffuse pollution from agricultural land using flow connectivity simulation. *Journal of Hydrology* **304**(1-4): 446-461.

Heathwaite, A.L., Billen, G., Gibson, C., Neal, C., Steenvoorden, J., Withers, P. and Bolton, L. (2005b) Nutrient mobility within river basins: a European Perspective. *Journal of Hydrology* **304** (1-4): 1-492.

Hegesh, E. and Shiloah, J. (1982) Blood nitrates and infantile methemoglobinemia. *Clinica Chimica Acta* **125**: 107-115.

Hering, D., Borja, A., Carstensen, J., Carvalho, L., Elliott, M., Feld, C.K., Heiskanen, A.S., Johnson, R.K., Moe. J., Pont, D., Solheim, A.L and De Bund, W.V. (2010) The European Water Framework Directive at the age of 10: A critical review of the achievements with recommendations for the future. *Science of the Total Environment* **408**(19): 4007-4019.

Herzog, F., Prasuhn, V., Spiess, E. and Richner, W. (2008) Environmental cross-compliance mitigates nitrogen and phosphorus pollution from Swiss agriculture. *Environmental Science and Policy* **II**: 655-668.

Hopkins, A. (2000) Herbage production. In: Hopkins, A. (Eds) *Grass, Its Production and Utilisation.* Blackwell Science, Oxford, UK:, pp. 90-110.

Howden, N.J.K. and Burt, T.P. (2008) Temporal and spatial analysis of nitrate concentrations from the Frome and Piddle catchments in Dorset (UK) for water years 1978-2007: Evidence for nitrate breakthrough? *Science of the Total Environment*. **407**: 507-526.

Howden, N.J.K., Bowes, M.J., Clark, A.D.J., Humphries, N. and Neal, C. (2009) Water quality, nutrients and the European union’s Water Framework Directive in a lowland agricultural region: Suffolk, south-east England. *Science of the Total Environment* **407**: 2966-2979.

Hristov, A.N., Hazen, W. and Ellsworth, J.W (2006) Efficiency of use of imported nitrogen, phosphorus and potassium and potential for reducing phosphorus imports on Idaho dairy farms. *Journal of Dairy Science* **89**: 3702-3712.

Hughes, A., Chilton, J., and Williams, (2006) A. Review and categorisation of nitrate transport in groundwater systems. Appendix V of ‘Investigating the Effectiveness of NVZ Action programme Measures: Development of a strategy for England’ (Lord et al., 2007). Report for Defra project WT03017.

Hunt, D.T.E., Dee, A.S. and Oakes, D.B. (2004). Updating the estimates of the source apportionment of N to UK waters. Phase 2. Prepared for Defra. Available at: http://www.fwr.org/defrawqd/wqd0002.htm (Accessed 8 September 2010).

Hutchins, M., Fezzi, C., Bateman, I., Posen, P. and Deflandre re-Vlandas, A. (2009) Cost-effective mitigation of diffuse pollution: setting criteria for river basin management at multiple locations. *Environmental Management* **44**: 256-267.

Hutson, J.L., Pitt, R.E., Koelsch, R.K., Houser, J.B. and Wagenet (1998) Improving dairy farm sustainability II: Environmental Losses and Nutrient Flows. *Journal of Production Agriculture* **11**(2): 233-239.

Iital, A., Stalnacke, P., Deelstra, J., Loigu, E. and Pihlak, M. (2005) Effects of large-scale changes in emissions on nutrient concentrations in Estonian rivers in the Lake Peipsi drainage basin. *Journal of Hydrology* **304**(1-4): 261-273.

Integrated Pollution Prevention and Control Directive (IPPC) (96/61/EC replaced by 2008/1/EC) *Official Journal of the European Communities* **L257**: 26-40 replaced by *Official Journal of the European Communities* **L24**: 1-22.

Jansons, V., Busmanis, P., Dzalbe, I. and Kristeina, D. (2003) Catchment and drainage field nitrogen balances and nitrogen loss in three agriculturally influenced Latvian watersheds. *European Journal of Agronomy* **20**: 173-179.

Johnes, P.J. (1996) Evaluation and management of the impact of land use change on the nitrogen and phosphorus load delivered to surface waters: The export coefficient modelling approach. *Journal of Hydrology* **183**(3-4): 323-349.

Johnes, P.J., Foy, R., Butterfield, D. and Haygarth, P.M. (2007) Land use scenarios for England and Wales: evaluation of management options to support good ecological status in surface freshwaters. Soil Use and Management **23**: 176-194.

Johnson, P.A. and Smith, P.N. (1996) The effects of nitrogen fertiliser rate, cultivation and straw disposal on the nitrate leaching from a shallow limestone soil cropped with winter barley. *Soil Use and Management* **12**: 67-71.

Johnson, P.A., Shepherd, M.A., Smith, P.N. (1997) The effect of crop husbandry and nitrogen fertiliser on nitrate leaching from a shallow limestone soil growing a five course combinable crop rotation. *Soil Use and Management* **13**: 17-23.

Johnson, P.A., Shepherd, M.A., Hatley, D.J. and Smith, P.N. (2002) Nitrate leaching from a shallow limestone soil growing a five course combinable crop rotation: the effects of crop husbandry and nitrogen fertiliser rate on losses from the second complete rotation. *Soil Use and Management* **18**:68-76.

Jurgen, S., Arno, L., Walter, K. (2006) Nutrient flows in suckler farm systems under two levels of intensity. *Nutrient cycling in agro-ecosystems* **74**: 41-57.

Kallis, G. and Butler, D. (2001): The EU water framework directive: measures and implications. *Water Policy* **3**: 125-142.

Kelly, J.R. (2001) Nitrogen Effects in Coastal Marine Ecosystems. In: Follett, R.F. and Hatfield, J.L. (Eds.) *Nitrogen in the environment: Sources, problems and management.* Elsevier, Amsterdam, NL., p207 – 255.

Kelm, M., Loges, R., Taube, F. (2008) Comparative analysis of conventional and organic farming systems: Nitrogen surpluses and nitrogen losses. In: Neuhoff, D., Halberg, N., Alfoldi, T., Lockeretz, W., Thommen, A., Rasmussen, I.A., Hermansen, J., Vaarst, M., Lueck, L., Caporalli, F., Jensen, H.H., Migliorini, P. and Willer, H. (Eds.) *Proceedings of the 16th International Federation of Organic Agriculture Movements Organic World Congress*, 16-20 June 2008, Modena, Italy.

Kersebaum, K.C., Steidl, J., Bauer, O.,and Piorr, H.P. (2003) Modelling scenarios to assess the effects of different agricultural management and land use options to reduce diffuse nitrogen pollution into the river Elbe. *Physics and Chemistry of the Earth* **28** (12-13): 537-545.

Kohn, R.A., Dou, Z., Ferguson, J.D., Boston, R.C. (1997) A sensitivity analysis of nitrogen losses from dairy farms. *Journal of environmental management* **50**: 417-428.

Kopke, U. (1987) Symbiotische Stickstoff Fixierung und Vorfuchtirkung von Ackerbohnen (*Vivia Faba*). Habilitation Thesis University of Gottingen. In: Watson, C.A., Bengtsson, H., Ebbesvik, M., Loes, A.K., Myrbeck, A., Salomon, E., Schroder, J. and Stockdate, E.A. (2002) A review of farm-scale nutrient budgets for organic farms. *Soil Use and Management* **18**: 264-273.

Korsaeth, A. and Eltun, R. (2000) Nitrogen mass balances in conventional, integrated and ecological cropping systems and the relationship between balance calculations and nitrogen runoff in an 8-year field experiment in Norway. *Agriculture, Ecosystems and Environment* **79**: 199-214.

Kristensen, P. and Hansen, O.H. (1994) European Rivers and Lakes – Assessment of their Environmental State, European Environment Agency, Environmental Monographs1, 122p.

Kristensen, E.S., Hogh-Jensen, H. and Kristensen, I.S. (1995) A simple model for estimation of atmospheric derived nitrogen in grass-clover systems. *Biological Agriculture and Horticulture* **12**:263-276.

Kronvang, B., Ertebjerg, G., Grant, G., Kristensen, R., Hovmand, P. and Kirkegaard, J. (1993) Nationwide monitoring of nutrients and their ecological effects: state of the Danish aquatic environment. *Ambio* **22**: 176-187.

Kronvang B, Jeppesen E, Conley DJ, Sondergaard M, Larsen SE, Ovesen NB, Carstensen J (2005). Nutrient pressures and ecological responses to nutrient loading reductions in Danish streams, lakes and coastal waters. *Journal of Hydrology* **304**(1-4): 274-288.

Kronvang, B., Andersen, H.E., Borgesen, C., Dalgaard, T., Larsen, S.E., Bogestrand, J. and Blicher-Mathiasen, G. (2008) Effects of policy measures implemented in Denmark on nitrogen pollution of the aquatic environment. *Environmental Science and Policy* **II** 144-152.

Kuipers. A.. and Mandersloot, F. (1999) Reducing nutrient losses on dairy farms in The Netherlands. *Livestock Production Science* **61**:139-144.

Kyllingsbaek, A. and Hansen, J.F. (2007) Development in nutrient balances in Danish agriculture 1980-2004. *Nutrient Cycling in Agro-ecosystems* **79**: 267-280.

Lambert, R., Sauvenier, X., Mathot, M. and Peeters, A. (2005) Evaluation of farm gate nitrogen balance. What is reachable considering nitrogen use efficiency? In: Schroder, J.J. and Neetson, J.J. (Eds.) *Proceedings of the 14th Nitrogen Workshop – N management in agrosystems in relation to the Water Framework Directive*. October 2005, Maastricht, the Netherlands, p83-85.

Langeveld, J.W.A., van Keulen, H., de Haan, J.J., Kroonen-Backbier, B.M.A. and Oenema, J. (2005) The nucleus and pilot farm research approach: experiences from The Netherlands. *Agricultural systems* **84**: 227-252.

Langeveld, J.W.A., Verhagen, A., Neeteson, J.J., van Keulen, H., Conijn, J.G., Schils, R.L.M. and Oenema, J. (2007) Evaluating farm performance using agri-environmental indicators: Recent experiences for nitrogen management in The Netherlands. *Journal of Environmental Management* **82**: 363-376.

Laws, J.A., Pain, B.F., Jarvis, S.C. and Scholefield, D. (2000) Comparison of grassland management systems for beef cattle using self contained farmlets: effects of contrasting nitrogen inputs and management strategies on nitrogen budgets, and herbage and animal production. *Agriculture, Ecosystems and Environment* **80**: 243-254.

Leach, K.A. and Bax, J.A. (1999) Efficiency of nitrogen use in dairy systems. In: Corrall, A. (Ed.) *Accounting for Nutrients – A Challenge for Grassland Farmers in the 21st Century.* British Grassland Society Occasional Symposium, Great Malvern, UK, **33**: 69-74.

Ledgard, S.F., Penno, J.W. and Sprosen, M.S. (1997) Nitrogen balances and losses on intensive dairy farms. Proceedings of the New Zealand Grassland Association **59**: 49-53.

Ledgard, S.F., Penno, J.W. and Sprosen, M.S. (1999) Nitrogen inputs and losses from clover / grass pastures grazed by dairy cows, as affected by nitrogen fertiliser application. *Journal of Agricultural Science*. **132:** 215-225.

Ledgard, S.F., Journeaux, H., Furness, H., Petch, R.A. and Wheeler, D,M. (2004) Use of nutrient budgeting and management options for increasing nutrient use efficiency and reducing environmental emissions from New Zealand farms. In:

Proceedings of OECD expert meeting on farm management indicators and the environment. 8-12 March 2004, Palmerston, New Zealand, 2004.

L'hirondel, J. and L'hirondel, J-L. (2002). *Nitrate and man. Toxic, harmless or beneficial?* CABI Publishing, Wallingford, UK.

Limbrick, K.J. (2003) Baseline nitrate concentration in groundwater of the Chalk in south Dorset, UK. *Science of the Total Environment* **314-316**: 89-98.

Liu, X., Ju, X., Zhang, F., Pan, J. and Christie, P. (2003) Nitrogen dynamics and budgets in a winter-wheat-maize cropping system in the North China Plain. *Field Crops Research* **83**(2): 111-124.

Liu, S.M., Brazier, R., and Heathwaite, A.L. (2005) An investigation into the inputs controlling predictions froth a diffuse phosphorus loss model for the UK; the Phosphorus Indicators Tool (PIT). *Science and the Total Environment* **344**(1-3): 211-223.

Loges, R., Kelm, M. and Taube, F. (2009) Comparative analysis of nitrogen surpluses and nitrogen losses of conventional and organic farms in Northern Germany In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.485-6

Lord, E.I. (1992) Modelling of nitrate leaching. *Aspects of Applied Biology* **30**: 19-28.

Lord, E.I. and Shepherd, M.A. (1993) Developments in the use of porous ceramic cups for measuring nitrate leaching. *Journal of Soil Science* **44**: 435-449.

Lord. E.I. 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., Johnson, P.A., and Archer, J.R. (1999) Nitrate Sensitive Areas: a study of large scale control of nitrate loss in England. *Soil Use and Management* **15**: 201-207.

Lord, E.I. and Anthony, S.G. (2000): MAGPIE: A modelling framework for evaluating nitrate loses at national and catchment scales. *Soil Use and Management* **16**: 167-174.

Lord, E.I., Anthony, S.G. and Goodlass, G. (2002) Agricultural nitrogen balance and water quality in the UK. Soil Use Management **18**(4): 363-369.

Lord, E., Shepherd, M., Silgram, M., Goodlass, G., Gooday, R., Anthony, S.G., Davison, P. and Hodgkinson, R. (2007) Investigating the effectiveness of NVZ Action Programme measures: Development of a strategy for England. Report for DEFRA project WT03017, 108pp. Available at: http://randd.defra.gov.uk/ (Accessed 26 August 2010).

LNV (Dutch Ministry of Agriculture, Nature and Food Quality). (2004) Third Dutch Action Programme (2004-2009) Concerning the Nitrates Directive (91/676/EEC). Ministerie van Landbouw, Natuur en Voedselkwaliteit, Den Haag, 2004.

Mabon, F., Raimbault, T., Moreau, P., Ruiz, L., Durand, P., Delaby, L. and Vertes, F. (2009) Meeting technical, economic and environmental efficiency in vulnerable areas. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.387-88.

Macdonald, A.J., Poulton, P.R., Powlson, D.S. and Jenkinson, D.S. (1997) Effects of season, soil type and cropping on recoveries, residues and losses of 15N-lablled fertiliser applied to arable crops in spring. *Journal of Agricultural Science*. **129**: 125-154.

Met Office (2010a) *UK climate averages - Yeovilton 1971-2000 averages* (online). Available at http://www.metoffice.gov.uk/climate/uk/averages/19712000/sites/yeovilton.html (Accessed 27 August 2010)

Met Office (2010b) *UK mapped climate averages - Annual average rainfall 1971-2000* (online). Available at http://www.metoffice.gov.uk/climate/uk/averages/regmapge.html (Accessed 27 August 2010)

Misselbrook, T.H., Van Der Weerden, T.J., Pain, B.F., Jarvis, S.C., Chambers, B.J., Smith, K.A., Phillips, V.R. and Demmers, T.G.M. (2000) Ammonia emission factors for UK agriculture. *Atmospheric Environment* **34**: 871-880.

Misselbrook, T.H., Chadwick D.R., Chambers, B.J., Smith, K.A., Webb, J., Demmers, T, and Sneath, R.W. (2006). Inventory of ammonia emissions from UK agriculture in 2004. Report for Defra project AM0127. Available at: http://randd.defra.gov.uk/ (Accessed 26 August 2010).

Mitchell, R.D.J., Harrison, R., Russell, K.J. and Webb, J. (2000) The effect of crop residue incorporation date on soil inorganic nitrogen, nitrate leaching and nitrogen mineralisation. *Biology and Fertility of Soils* **32**: 294-301.

Molenat, J. and Gascuel-Odoux, C. (2002) Modelling flow and nitrate transport in groundwater for the prediction of water travel times and of consequences of land use evolution on water quality. *Hydrological Processes* **16**(2): 479-492.

Mulier, A.., Hofman, G., Baecke, E., Carlier, L., De Brabander, D., De Groote, G., De Wilde, R., Fiems, L., Janssens, G., Van Cleemput, O., Van Herck, A., Van Huylenbroeck, G., and Verbruggen, I. (2003).. A methodology for the calculation of farm level nitrogen and phosphorus balances in Flemish agriculture. *European Journal of Agronomy* **20**: 45-51.

Natural England (2010) *Environmental Stewardship* (online). Available at: http://www.naturalengland.org.uk/ourwork/farming/funding/es/default.aspx (Accessed 25 August 2010).

Nevens, F., Verbruggen, I., Reheul, D. and Hofman, G. (2006) Farmgate nitrogen surpluses and nitrogen use efficiency of specialised dairy farms in Flanders: Evolution and future goals. *Agricultural Systems* **88**: 142-155.

Newmann, J.R., Anderson, N.J., Bennion, H., Bowes, M.J., Carvalho, L., Dawson, F.H., Furse, M., Gunn, I., Hilton, J., Hughes, R., Johnston, A.M., Jones, J.I., Luckes, S., Maitland, P., May, L., Monteith, D., O'Hare, M., Taylor, R., Trimmer, M. and Winder, J. (2005) Eutrophication in Rivers: An Ecological Perspective. Appendix VI of 'Investigating the Effectiveness of NVZ Action programme Measures:

Development of a strategy for England' (Lord et al., 2007). Report for DEFRA project WT03017, 2006, 37pp. Available at http://randd.defra.gov.uk/ (Accessed 26 August 2010).

Nielsen, A.H. and Kristensen, I.S. (2005) Nitrogen and phosphorous surpluses on Danish dairy and pig farms in relation to farm characteristics. *Livestock Production Science* **96**: 97-107.

Nissen, T.M., and Wander, M.M. (2003) Management and soil quality effects on fertiliser-use efficiency and leaching. *Soil Science Society of America Journal* **67**: 1524-1532

Nitrates Directive (91/676/EEC) *Official Journal of the European Communities* **L375**: 1–8.

Oborn, I., Edwards, A.C., Witter, E., Oenema, O., Ivarsson, K., Withers, P.J.A., Nilsson, S.I. and Stinzing, A.R. (2003) Element balances as a tool for sustainable nutrient management: a critical appraisal of their merits and limitations within an agronomic and environmental context. *European Journal of Agronomy* **20**(1-2): 211-225.

Oenema, O., Boers, P.C.M., van Eerdt, M.M., Fraters, B., van der Meer, H.G., Roest, C.W.J., Schroder, J.J. and Willems, W.J. (1998) Leaching of nitrate from agriculture to groundwater: the effect of policies and measures in the Netherlands. Environmental Pollution **102**: 471-478.

Oenema, J., Koskamp, G.J., Galama, P.J. (2001) Guiding commercial pilot farms to bridge the gap between experimental and commercial dairy farms; the project cows and opportunities. *Netherlands Journal of Agricultural Science* **49**: 277-296.

Oenema, O., Kros, H. and de Vries, W. (2003) Approaches and uncertainties in nutrient budgets: implications for nutrient management and environmental policies. European Journal of Agronomy **20**(1-2): 3-16.

Oenema, O., van Liere, L. and Schoumans, O. (2005) Effects of lowering nitrogen and phosphorus surpluses in agriculture on the quality of groundwater and surface water in the Netherlands. *Journal of Hydrology* **304**(1-4):289-301.

Ondersteijn, C.J.M., Beldman, A.C.G., Daatselaar, C.H.G., Giesen, G.W.J. and Huirne, R.B.M. (2002) The Dutch Mineral Accounting System and the European Nitrate Directive: implications for N and P management and farm performance. *Agriculture Ecosystems and Environment* **92**(2-3): 283-296.

Organisation for Economic Co-operation and Development (OECD) (1982). Eutrophication of Waters; monitoring, assessment and control. OECD Paris, Tech. Report F 52/11.50. 153pp.

Organisation for Economic Co-operation and Development (OECD) (2002) Gross Nitrogen Balance Handbook, OECD / EUROSTAT, Paris.

Oslo and Paris Conventions for the Prevention of Marine Pollution Commission (OSPARCOM) (1994) Guidelines for Calculating Mineral Balances, Working Group on Nutrients, NUT 94/8/1-E, Berne, Switzerland, 1994.

Owens, N.J.P. (1993) Nitrate Cycling in marine waters. In: Burt, T.P., Heathwaite, A.L. and Trudgill, S.T. (Eds.) *Nitrate: processes, patterns and management*. Wiley, Chichester, UK. p444.

Powell, J.M., Rotz, C.A., Weaver, D.M. (2009) Nitrogen use efficiency in dairy production. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.241-2.

Power, I., Ledgard, S. and Monaghan, R. (2002) Nutrient Budgets for Three Mixed Farming Catchments in New Zealand. Ministry of Agriculture and Forestry Technical Paper No.17 pp42.

Quinn, P. (2004) Scale appropriate modelling: representing cause and effect relationships in nitrate pollution at the catchment scale for the purpose of catchment scale planning. *Journal of Hydrology* **291**: 197-217.

Radach, G., Berg, J. and Hagmeier, E. (1990) Long term changes in the annual cycles of meteorological, hydrgraphic, nutrient and phytoplankton time series at

Heligoland and at LV Elbe 1 in the German Bight. *Continental Shelf Research* **10**: 305-328.

Reidy, B., Uebersax, A. and Menzi, H. (2005) Nitrogen and phosphate surpluses after the introduction of a new agricultural policy in Switzerland. In: Schroder, J.J. and Neetson, J.J. (Eds.) *Proceedings of the 14th Nitrogen Workshop – N management in agrosystems in relation to the Water Framework Directive.* October 2005, Maastricht, the Netherlands, p98-100.

Roberts, D.J., Leach, K.A. and Goldie, J. (2007) Assessment and improvement of the efficiency of nitrogen use on commercial dairy farms. *International Journal of Agricultural Sustainability* **5**: 295-304.

Rotz, C.A., Taube, F., Russelle, M.P., Oenema, J., Sanderson, M.A. and Wachendorf, M. (2005) Whole farm perspectives of nutrient flows in grassland agriculture. *Crop Science* **45** (6) 2139-2159.

Ruiz, L., Abiven, S., Durand, P., Martin, C., Vertes, F. and Beaujouan, V. (2002) Effect on nitrate concentration in stream water of agricultural practices in small catchments in Brittany: I Annual nitrogen budgets. *Hydrology and Earth System Sciences* **6**(3) 497-505.

Rukin, N., Roy, S. and Davison, P. (2008) Modelling nitrate concentrations with variation in time. Annex 53-UK to Final WAgriCo report. Available at http://www.wagrico.org/content/default.asp?PageId=231&LanguageId=0 (Accessed 26 August 2010).

Runge, T. and Osterburg, B. (2007) A results-oriented approach to reward N-efficiency improvements. In: *Proceedings of the 15th Nitrogen Workshop*, 28-30 May 2007 Lleida, Spain.

Sacco, D., Bassanino, M. and Grignani, C. (2003) Developing a regional agronomic information system for estimating nutrient balances at a larger scale. *European Journal of Agronomy* **20**: 199-210.

Salo, T. and Turtola, E. (2006) Nitrogen balance as an indicator of nitrogen leaching in Finland. *Agriculture, Ecosystems and Environment* **113**: 98-107.

Schmidt, H., Phillips, L., Welsh, J.P., and von Fragstein, P. (1999) Legume breaks in stockless organic farming rotations: Nitrogen accumulation and influence on the following crops. *Biological Agriculture and Horticulture* **17**: 159-170.

Schroder, J.J., Aarts, H.F.M., ten Berge, H.F.M., van Keulen, H. and Neetson, J.J. (2003). An evaluation of whole farm nitrogen balances and related indices for efficient nitrogen use. *European Journal of Agronomy* **20**: 33-44.

Schroder, J.J., Scholefield, D., Cabral, F. and Hofman, G. (2004) The effects of nutrient losses from agriculture on ground and surface water quality: the position of science in developing indicators for regulation. *Environmental Science and Policy* **7**(1):15-23.

Shepherd, M.A., Davis, D.B. and Johnson, P.A.. (1993) Minimising nitrate losses from arable soils. *Soil Use and Management* **9**(3): 94-99.

Shepherd, M.A., Stockdale, E., Powlson, D. and Jarvis, S. (1996) The influence of organic nitrogen mineralisation on the management of agricultural systems in the UK. *Soil Use and Management* **12**: 76-85.

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

Shepherd, M.A. and Webb, J. (1999) Effects of overwinter cover on nitrate loss and drainage from a sandy soil: consequences for water management? *Soil Use and Management* **15**: 109-116.

Shepherd, M.A., Hatch, D.J., Jarvis, N.J. and Bhogal, A. (2001) Nitrate leaching from reseeded pasture. *Soil Use and Management* **17**: 97-105.

Shepherd, M. and Chambers, B. (2007) Managing nitrogen on the farm: the devil is in the detail. *Journal of the Science of Food and Agriculture* **87**: 558-568.

Shepherd, M., Wheeler, D. and Power, I. (2009) The scientific and operational challenges of using an OVERSEER nutrient budget model within a regulatory framework to improve water quality in New Zealand. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.601-2.

Sieling, K., Gunther-Borstel, O. and Hanus, H. (1997) Effect of slurry application and mineral nitrogen fertilisation on N leaching in different crop combinations. *Journal of Agricultural Science* **128**: 79-86.

Sieling, K., Schroder, H. and Hanus, H. (1998a) Mineral and slurry nitrogen effects on yield, N uptake and apparent N-use efficiency of oilseed rape (brassica napus). *Journal of Agricultural Science* **130**: 165-172.

Sieling, K., Schroder, H., Finck, M. and Hanus, H. (1998b) Yield, N uptake, and apparent N-use efficiency of winter wheat and winter barley grown in different cropping systems. *Journal of Agricultural Science* **131**: 375-387.

Sieling, K. and Kage, H. (2006) N balance as an indicator of N leaching in an oilseed rape – winter wheat – winter barley rotation. *Agriculture, Ecosystems and Environment* **115**: 261-269.

Silgram, M. and Shepherd, M. (1999) The effect of cultivation on soil nitrogen mineralisation. *Advances in Agronomy* **65**: 267-311.

Silgram, M. and Chambers, B.J. (2002) Effects of long-term straw management and fertiliser additions on soil nitrogen supply and crop yields at two sites in eastern England. *Journal of Agricultural Science* **139**:115-127.

Silgram, M., Williams, A., Waring, R., Neumann, I., Hughes, A., Mansour, M. and Besien, T. (2005) Effectiveness of the nitrate sensitive areas scheme in reducing groundwater concentrations in England. Quarterly Journal of Engineering Geology and Hydrogeology **38**: 117-127.

Skeffington, R. (2002) European nitrogen policies, nitrate in rivers and the use of the INCA model. *Hydrology and Earth System Sciences* **6**(3): 315-324.

Smith, V.H., Tilman, G.D. and Nekola, J.C. (1999) Eutrophication: impacts of excess nutrient inputs on freshwater, marine and terrestrial ecosystems. *Environmental Pollution* **100**: 179-196.

Smith, K.A., Jackson, D.R. and Pepper, T.J. (2001) Nutrient losses by surface run-off following the application of organic manures to arable land. 1. Nitrogen. *Environmental Pollution* **112**: 41-51.

Smith, K.A., Beckwith, C.P., Chalmers, A.G. and Jackson, D.R. (2002) Nitrate leaching following autumn and winter applications of animal manures to grassland. *Soil Use and Management* **18**: 428-434.

Smith, K.A. and Cottrill, B.R. (2007) Nitrogen output of livestock excreta. Final report of DEFRA project WT0715NVZ. Available at: http://randd.defra.gov.uk/ (Accessed 26 August 2010).

Sonneveld, M.P.W. and Bos, J.F.F.P. (2009) N management strategies on Dutch dairy farms in relation to ammonia losses. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.499-500.

Sorensen, P. and Thomsen, I.K. (2009) Transformations and plant availability of animal manure N – a review. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.229-230.

Stalnacke, P., Grimvall, A., Libiseller, C., Laznik, A. and Kokorite, I. (2003) Trends in nutrient concentrations in Latvian rivers and the response to the dramatic change in agriculture. Journal of Hydrology **283**(1-4): 184-205.

Stokes, D.T., Scott, R.K., Tilston, C.H., Cowie, G. and Sylvester Bradley, R. (1992) The effect of time of soil disturbance on nitrate mineralisation. *Aspects of Applied Biology (Nitrate and Farming Systems)* **30**: 279-282

Swensson, C. (2003) Analysis of mineral element balances between 1997 and 1999 from dairy farms in the south of Sweden. European Journal of Agronomy **20**: 63-69.

Sylvester-Bradley, R. and Cross, R.B. (1991) Nitrogen residues from peas and beans and the response of the following cereal to applied nitrogen. *Aspects of Applied Biology* **27**: 293-298.

Sylvester-Bradley, R. (1993) Scope for more efficient use of fertiliser nitrogen. *Soil Use and Management* **9**(3): 112-117.

Sylvester-Bradley, R. and Kindred, D.R. (2009) Analysing nitrogen responses of cereals to prioritise routes to the improvement of nitrogen use efficiency. *Journal of Experimental Botany* **60**(7): 1939-1951.

Tannenbaum, S.R. (1987) Endogenous formation of N-nitroso compounds: a current perspective. In: Bartsch, H., O'Neil, I.K. and Schulte-Hermann, R. (Eds.) *Relevance of N-nitroso compounds to human cancer: exposure and mechanisms,* IARC Scientific Publications 84, Lyon, pp 292-298.

Torstensson, G. and Aronsson, H. (2000) Nitrogen leaching and crop availability in manured catchment cop systems in Sweden. Nutrient Cycling in Agro-ecosystems. **56**: 139-152.

Treacy, M., Humphreys, J., McNamara, K., Browne, R. and Watson, C.J. (2008). Farmgate nitrogen balances on intensive dairy farms in the southwest of Ireland. Irish *Journal of Agricultural and Food Research* **47**(2):105-117.

UKTAG (2010): *UKTAG Biological Assessment Methods* (online). Available at: https://www.wfduk.org/bio_assessment (Accessed 26 August 2010).

Ulen, B.M. and Kalisky, T. (2005) Water erosion and phosphorus problems in an agricultural catchment - Need for natural research for implementation of the EU Water Framework Directive. *Environmental Science and Policy* **8**: 477- 484.

UNESCO (2007) *Frome-Piddle (UK) HELP catchment* (online). Available at: http://portal.unesco.org/science/fr/ev.php-URL_DO=DO_PRINTPAGE&URL_SECTION=201.html (Accessed 27 August 2010).

Urban Waste Water Treatment Directive (91/271/EEC) *Official Journal of the European Communities* **L135**: 40-52. Amended to clarify standards for total N and P by directive 98/15/EEC *Official Journal of the European Communities* **L67**: 29-30.

Vagstad, N., Stalnacke, P., Andersen, H.E., Deelstra, J., Jansons, V., Kyllmar, K., Loigu, E., Rekolainen, S. and Tumus, R. (2004) Regional variations in diffuse nitrogen losses in the Nordic and Baltic regions. *Hydrology and Earth System Sciences* **8**(4): 651-662.

Van Beek, C.L., Brouwer, L. and Oenema, O. (2003) The use of farmgate balances and soil surface balances as estimator for nitrogen leaching to surface water. *Nutrient Cycling in Agro-ecosystems* **67**(3): 233-244.

Van Bruchem, J., Schiere, H. and van Keulen, H. (1999) Dairy farming in the Netherlands in transition towards more efficient nutrient use. *Livestock Production Science* **61**: 145-153.

Van der Werf, H.M.G. and Petit, J. (2002) Evaluation of the environmental impact of agriculture at the farm level: a comparison and analysis of 12 indicator-based methods. *Agriculture, Ecosystems and Environment* **93**: 131-145.

Van Eerdt, M.M. and Fong, P.K.N. (1998) The monitoring of nitrogen surpluses from agriculture. *Environmental Pollution* **102** S1 227-233.

Van Lanen, H.A.J. and Dijksma, R. (1999) Water flow and nitrate transport to a groundwater-fed stream in the Belgian-Dutch chalk region*. Hydrological Processes* **13**(3): 295-307.

Van Loon, A.J., Botterweck, A.A., Goldbohm, R.A., Brants, H.A., Van Klaveren, J.D. and Van den Brandt, P.A. (1998). Intake of nitrate and nitrite and the risk of gastric cancer: a prospective cohort study. *British Journal of Cancer* **78**: 129-135.

Van Wepern, W. (2009) Making the flip: Dutch dairy farmers reconnect with the soil. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting* *different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.609-610.

Verbruggen, I., Nevens, F., Mulier, A., Reheul, D. and Hofman, G. (2005) Nitrogen surpluses of Flemish dairy, beef and arable farms: changes between 1992-2000. In: Schroder, J.J. and Neetson, J.J. (Eds.) *Proceedings of the 14th Nitrogen Workshop – N management in agrosystems in relation to the Water Framework Directive*. October 2005, Maastricht, the Netherlands, p111-113.

Vinten, A.J.A. and Dunn, S.M. (2001) Assessing the effects of land use on temporal change in well water quality in a designated nitrate vulnerable zone. *Science of the Total Environment* **265**: 253-268.

Vinther, F.P., Borgesen, C.D. and Waagepetersen, J. (2009) Reduction of N surplus and leaching: restrictions and possibilities. In: Grignani, C., Acutis, M., Zavattaro, L., Bechini, L., Bertora, C., Marino Gallina, P., and Sacco, D. (Eds.) *Proceedings of the 16th Nitrogen Workshop – Connecting different scales of nitrogen use in agriculture.* 28 June – 1 July 2009 Turin, Italy, p.611-2.

Vos, J. and van der Putten, P.E.L. (2004) Nutrient cycling in a cropping system with potatoes, spring wheat, sugar beet, oats and nitrogen catch crops. II. Effect of catch crops on nitrate leaching in autumn and winter. *Nutrient Cycling in Agro-ecosystems* **70**: 23-31.

Wade, A.J., Durand, P., Beaujouan, V., Wessel, W.W., Raat, K.J., Whitehead, P.G., Butterfield, D., Rankinen, K. and Lepisto, A. (2002) A nitrogen model for European catchments: INCA, new model structure and equations. *Hydrology and Earth System Sciences* **6**(3): 559-582.

Wade, A.J., Neal, C., Butterfield, D. and Futter, M.N. (2004) Assessing nitrogen dynamics in European ecosystems, integrating measurement and modelling: conclusions. *Hydrology and Earth System Sciences* **8**(4): 846-857.

WAgriCo (2006) *Water Resources Management in Cooperation with Agriculture – Study catchment areas* (online). Available at: http://www.wagrico.org/content/default.asp?PageId=183% LanguageId=0 (Accessed 27 August 2010).

WAgriCo (2008) Results of the collected farm data and concept for model farm measurement network – Annex 45 to WAgriCo final report. Available at: http://www.wagrico.org/content/default.asp?PageId=231&LanguageId=0 (Accessed 26 August 2010).

Wahlin, K. and Grimvall, A. (2008) Uncertainty in water quality data and its implications for trend detection: lessons from Swedish environmental data. *Environmental Science and Policy* **11**(2): 115-124.

Water Framework Directive (2000/60/EC) *Official Journal of the European Communities* **L327**: 1-72.

Watson, C.A. and Atkinson, D. (1999) Using nitrogen budgets to indicate nitrogen use efficiency and losses from whole farm systems: a comparison of three methodological approaches. *Nutrient cycling in Agro-ecosystems* **53**: 259-267.

Webb, J., Sylvester-Bradley, R. and Seeney, F.M. (1997) The effects of site and season on the fate of nitrogen residues from root crops grown on sandy soils. *Journal of Agricultural Science* **128**: 445-460.

Webb, J., Seeney, F.M. and Sylvester-Bradley, R. (1998). The response to fertiliser nitrogen of cereals grown on sandy soils. *Journal of Agricultural Science* **130**: 271-286.

Webb, J., Harrison, R. and Ellis, S. (2000) Nitrogen fluxes in three arable soils in the UK. *European Journal of Agronomy* **13**: 207-223.

Webb, J. (2001) Estimating the potential for ammonia emissions from livestock excreta and manures. *Environmental Pollution* **III**: 395-406.

Webb, J., Ellis, S., Harrison, R. and Thorman, R. (2004) Measurement of N fluxes and soil N in two arable soils in the UK. *Plant and Soil* **260**: 253-270.

Webster, C.P., Shepherd, M.A., Goulding, K.W.T. and Lord, E.I. (1993) Comparisons of methods for measuring the leaching of mineral nitrogen from arable land. *Journal of Soil Science* **44**: 49-62.

Wendland, F., Bogena, H., Goemann, H., Hake, J.F., Kreins, P. and Kunkel, R. (2005) Impact of nitrogen reduction measures on the nitrogen loads of the river Ems and Rhine (Germany). *Physics and Chemistry of the Earth* **30**: 527-541.

Wheeler, D.M., Ledgard, S.F., deKlein, C.A.M., Monaghan, R.M., Carey, P.L., McDowell, R.W. and Johns, K.L. (2003) OVERSEER nutrient budgets – moving towards on-farm resource accounting. *Proceedings of the New Zealand Grassland Association* **65**: 191-194.

Whitehead, P.G., Wilson, E.J. and Butterfield, D. (1998) A semi-distributed Integrated Nitrogen model for multiple source assessment in Catchments (INCA): Part I - model structure and process equations. *Science of the Total Environment* **210**(1-6): 547-558.

Williams, J.R. and Lord, E.I. (1997) The use of porous ceramic cup water samplers to measure solute leaching on chalk soils. *Soil Use and Management*. **13**: 156-162.

Withers, P.J.A. and Lord, E.I. (2002) Agricultural nutrient inputs to rivers and groundwaters in the UK: policy, environmental management and research needs. *Science of the Total Environment* **282**: 9-24.

Withers, P.J.A., Edwards, A.C. and Foy, R.H. (2001) Phosphorus cycling in UK agriculture and implications for phosphorus loss from soil. *Soil Use and Management* **17**(3): 139-149.

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.

Wriedt, G. and Rode, M. (2006) Modelling nitrate transport and turnover in a lowland catchment system. *Journal of Hydrology* **328**(1-2):157-17.

Zalidis, G.C., Tsiafouli, M.A.., Takavakoglou, V., Bilas, G. and Misopolinos, N. (2004) Selecting agri-environmental indicators to facilitate monitoring and assessment of EU agri-environmental measures effectiveness. *Journal of Environmental Management* **70**: 315-321.

Zwart, M.H., Hooijboer, A.E.J., Fraters, B., Kotte, M., Duin, R.N.M., Daatselaar, C.H.G., Olstoorn, C.S.M. and Bosma, J.N.M. (2008). Agricultural practice and water quality in the Netherlands in the 1992-2006 period. RIVM report produced for the National Institute for public health and the environment.

Appendix

Table A-1: The magnitude of soil surface / system budget components. Average values presented in table 5-3 to illustrate the relative magnitude of each flux and aid development of a nutrient budget for the evaluation of mitigation effectiveness.

Input / output	Flux	Reference	N type / Land use	Magnitude
Input	Atmospheric deposition	Goulding et al., 1998	General	rural areas = 30kg N ha^{-1}
Input	Atmospheric deposition	Goulding et al., 1998	General	urban areas = 40kg N ha^{-1}
			Mean	35.0 kg N ha^{-1}
Input	Fertiliser inputs	BSFP, 2009	Arable	Average application 2004 – 2008 = 147kg N ha^{-1}
Input	Fertiliser inputs	BSFP, 2009	Grassland	Average application 2004 - 2008 = 69kg N ha^{-1}
			Mean	95 kg N ha^{-1}
Input	Excreta	(NVZ regulations)	General	(Inc. within NVZ max application value below) – 832,000 tonnes total excreta N for England and Wales (Webb et al., 2001).
			Mean	n/a
Input	Manure inputs	NVZ regulations	General	NVZ max manure application =\170 total kg N ha^{-1}
			Mean	170.0 kg N ha^{-1}
Input	Mineralisation	Sylvester-Bradley, 1993	Arable	Typical mineralisation (soil supply) inputs to arable crops in UK 80kg N ha^{-1}
Input	Mineralisation	Grylls et al., 1997	Arable	Apparent mineralisation averaged 26kg N ha^{-1} on shallow soils over chalk in Southern England during growing season
Input	Mineralisation	Webb et al., 1997	Arable	Apparent mineralisation averaged 37 / 52- 62kg N ha^{-1} cereal after cereal / cereal after sugar or potatoes on sandy soils in England during growing season.
Input	Mineralisation	Webb et al., 1997	Arable	Total net mineralisation input (includes input over winter months) 80, 95, 115 kg N ha^{-1} following cereal, potatoes and sugar beet.
Input	Mineralisation	Webb et al., 2000	Arable	Apparent mineralisation averaged 51kg N ha^{-1} on sandy soils in England during growing season.
Input	Mineralisation	Goulding, 2000	General	Gross inputs ranged from approx. 0.6 - 3.3 mg N kg^{-1} day^{-1}. Arable < grass < 2yr ley < reseed. Clay loam > loam > sand
			Mean	66.0 kg N ha^{-1}

Input / output	Flux	Reference	N type / Land use	Magnitude
Input	N fixation	Cuttle et al., 2003	General	N fixation ranged from 105 kg N ha^{-1} (fresh peas) to 300kg N ha^{-1} (red clover)
Input	N fixation	Goulding, 1990	General	Free living soil bacteria = 5kg N ha^{-1}
Input	N fixation	Kopke (1987)	General	Spring / winter beans = 200kg N ha^{-1}
Input	N fixation	Kristensen et al., 1995	General	White clover = 150 / 85 kg N ha^{-1} for 1-2 and >2 yr leys
Input	N fixation	Schmidt et al., 1999	General	Red clover = 240kg N ha^{-1}
Input	N fixation	Sylvester-Bradley and Cross, 1991	General	Spring / winter bean residue = 25kg N ha^{-1}
			Mean	139.0 kg N ha^{-1}
Output	Ammonia volatilisation	Defra, 2002b	General	Ammonia losses were 10 fold greater than DN losses from FYM on arable land (nh3 loss @41% of N). On grassland ammonia losses were 3 times higher than DN losses (NH3 losses @ 23% total N)
Output	Ammonia volatilisation	MANNER (Chamber et al., 1999)	General	Spreading losses only
Output	Ammonia volatilisation	Aarts et al., 2000	Excreta	7% excreta N lost via ammonia volatilisation
Output	Ammonia volatilisation	Bussink, 1992	Excreta	250N sward, 3.1% excreta N lost as NH3
Output	Ammonia volatilisation	Misselbrook et al., 2006	Excreta	Cattle grazing = 5.1 and 1.6% TAN excreted for dairy and beef cattle respectively
Output	Ammonia volatilisation	Misselbrook et al., 2006	Excreta	Sheep grazing losses = 11 and 18% TAN for sheep and lambs
Output	Ammonia volatilisation	Webb et al., 2001	Excreta	8.04% excreted TAN lost via ammonia during grazing
			Mean	7.7% excreta N
Output	Ammonia volatilisation	Misselbrook et al., 2006	Housing/ storage	Housing losses cattle 31.4 / 33.2% TAN for cattle housed on slurry / straw.
Output	Ammonia volatilisation	Misselbrook et al., 2006	Housing/ storage	Also hard standing losses (cattle)
Output	Ammonia volatilisation	Misselbrook et al., 2006	Housing/ storage	Storage = 34.8% TAN (cattle)
Output	Ammonia volatilisation	Smith and Cottrill, 2007	Housing/storage	Housing and storage losses of excreted N
Output	Ammonia volatilisation	Misselbrook et al., 2006	FYM	FYM land application = 81% TAN lost
Output	Ammonia volatilisation	Misselbrook et al., 2006	Layer	Land application of poultry manure = 63% UAN
Output	Ammonia volatilisation	Misselbrook et al., 2006	Layer	storage losses 8.7% TAN
Output	Ammonia volatilisation	Misselbrook et al., 2006	Layer	Housing losses of 37.4 TAN

Input / output	Flux	Reference	N type / Land use	Magnitude
Output	Ammonia volatilisation	Misselbrook et al., 2006	Slurry	slurry land application = 15, 37 and 59% TAN for <4, 4-8 and >8% DM
Output	Ammonia volatilisation	Misselbrook et al., 2006	Slurry	Injection of cattle slurry = 70% reduction
Output	Ammonia volatilisation	Misselbrook et al., 2006	Slurry	Injection of pig slurry = 90% reduction
			Mean	Housing = 24% of manure N, storage = 15% of manure N, spreading = 36% of manure N not inc values from WT0751NVZ or MANNER
Output	Denitrification	Ledgard et al., 1999	Excreta	4% excreta N denitrified
Output	Denitrification	Webb et al., 2001	Excreta	N2O losses during grazing = 2%
			Mean	3.0% excreta N
Output	Denitrification	Defra, 2002b	Fertiliser	DN losses from inorganic fertiliser were on average 7.7% of the total N applied (range = 0.7 - 16.2 kg N ha^{-1})
Output	Denitrification	Ledgard et al., 1999	Fertiliser	Typically 2-10% fertiliser applied. However no accounting for soil type or drainage, and depends on fertiliser application rate.
Output	Denitrification	Webb et al., 2001	Fertiliser	NO losses approx 2.5% fertiliser N
			Mean	5.6% fertiliser N
Output	Denitrification	Conon et al., 2000	General	Daily DN figures which equate to 5.5 kg N ha^{-1} in cold, dry, low SMN conditions and kg N ha^{-1} in wet, warm, high SMN soil. For dry, warm, high SMN soils 91.25 kg N ha^{-1} can be estimated.
Output	Denitrification	Ledgard et al., 1999	General	3-7 kg N ha^{-1} on 0N farms and 30-70 kg N ha^{-1} on fertilised grass.
Output	Denitrification	Webb et al., 2001	General	N2O losses during land applications = 1.25%
Output	Denitrification	Webb et al., 2001	General	N2 losses are estimated as 3 times N2O losses
Output	Denitrification	Webb et al., 2005	General	On clay loam under conventional grazing DN losses ranged from 29 - 113 kg N ha^{-1}. For sandy loam figures ranged from 1-10 kg N ha^{-1}.
Output	Denitrification	Webb et al., 2001	General	0.3% TAN in manure and excreta
Output	Denitrification	Webb et al., 2001	FYM	N2O losses during storage = 2% FYM
Output	Denitrification	Webb et al., 2005	FYM	Losses = 18% ammoniacal N from FYM
Output	Denitrification	Defra, 2002b	Cattle FYM	Cattle FYM = 6.7% RAN
Output	Denitrification	Defra, 2002b	Cattle slurry	Cattle slurry = 8.6% RAN

Input / output	Flux	Reference	N type / Land use	Magnitude
Output	Denitrification	Defra, 2002b	Pig slurry	Pig slurry = 4.2% RAN
Output	Denitrification	Webb et al., 2001	Slurry	N2O losses during storage = 1% slurry
Output	Denitrification	Webb et al., 2005	Slurry	Losses = 13.2% of ammoniacal N from slurry??
Output	Denitrification	Defra, 2002b	Layer manure	Layer manure = 2.7% RAN
			Mean	3.8% manure N (based on % figures only)
Output	Leaching	Goulding, 2000	Arable	Losses from arable land under good nitrate practice range from 40 -70 kg N ha^{-1} for cereals, 80 – 110 kg N ha^{-1} for potatoes, 90 – 140 kg N ha^{-1} for peas and 120 -170 kg N ha^{-1} (approx. figures) from rotational set-aside
Output	Leaching	Johnson et al., 2002	Arable	49 kg N ha^{-1}lost on average from complete rotation under standard practices
Output	Leaching	Lord et al., 1999	Arable	NSA Baseline - 53 kg N ha^{-1}across all sites. Cereal = 30 kg N ha^{-1}, potatoes / sugar beet = 73 kg N ha^{-1}
Output	Leaching	WAgriCo measurements	Arable/grassland	2007 / 2008: grass = 49.5 / 82.5 kg N ha^{-1}, maize = 122.1 / 306.4 kg N ha^{-1}, winter wheat = 72.5 / 157.5 kg N ha^{-1}, SBM 58.4 / 161.8 kg N ha^{-1}, WOSR = 157.7 / 196 kg N ha^{-1}
Output	Leaching	Beckwith et al., 1998	General	Slurry applied = 55.5 kg N ha^{-1}
Output	Leaching	Beckwith et al., 1998	General	Broiler litter applied = 24.1 kg N ha^{-1}
Output	Leaching	Beckwith et al., 1998	General	FYM applied = 10.3 and 18.7 kg N ha^{-1} (dependant on site)
Output	Leaching	Berry et al., 2003	General	38 -136 kg N ha^{-1} estimated using NITCAT
Output	Leaching	Cuttle and Scholefield, 1995	Grassland	Losses from grassland range from 30 to >200 kg N ha^{-1}
Output	Leaching	Goulding, 2000	Grassland	Losses from grassland under good nitrate practice range from 40 - 110 kg N ha^{-1}
			Mean	Grassland = 85.3 (30 - > 200 kg N ha^{-1})
			Mean	Arable = 96.3 (10.3 - 306.4) kg N ha^{-1})

NOTES

TAN = 60% TN for cattle and sheep, and 70% TN for pigs and poultry

Table A-2: Inputs and outputs of European farm gate budget methodologies

Country	Details	Reference	Inputs									Outputs				
			Fertiliser	Feed	N fixation	Animals	Manure	Atmos. Depos.	Seeds	Bedding	Other	Animals	Animal products	Crop offtake	manure	Other
Belgium	120 farms	Nevens et al., 2006	x	x	x	x	x	x		x		x	x	x	x	
	National	Verbruggen et al., 2005	x	x	x	x	x	x		x		x	x	x		
	40 farms	Mulier et al., 2003	x	x		x	x			x		x	x	x	x	
Denmark	41 farms	Dalgaard et al., 2002	x	x	x	x	x	x	x			x	x	x	x	
	National	Kyllingsbaek and Hansen, 2007	x	x	x		x	x				x	x	x		
	Green accounts	Goodlass et al., 2003	x	x	x	x	x	x	x		Irrigation	x	x	x	x	
	Ethical account for livestock farms	Goodlass et al., 2003	x	x	x	x	x	x	x		Irrigation	x	x	x	x	
	75 farms	Nielsena and Kristensen, 2005	x	x	x	x	x	x	x	x	Store changes	x	x	x	x	
	20 farms	Halberg, 1999			x			x								
Germany	32 farms	Kelm et al., 2008	x	x	x	x						x	x	x	x	NH3
	32 farms	Loges et al., 2009	x	x	x	x						x	x	x	x	
Italy	41 farms	Bassanino et al., 2007	x	x	x	x		x			Roughage, litter	x	x	x	x	
Luxembourg	FHL herdbooks system	Goodlass et al., 2003	x	x		x	x		x		Irrigation, soil analysis	x	x	x	x	
Netherlands	17 farms (MINAS)	Oenema et al., 2001	x	x		x	x				Roughage	x	x		x	Gaseous losses
	194 farms (MINAS)	Ondersteijn et al., 2002	x	x		x	x				Roughage	x	x	x	x	
	National (MINAS)	Zwart et al., 2008	x	x				x				x	x	x	x	NH3

Country	Details	Reference	Inputs									Outputs				
			Fertiliser	Feed	N fixation	Animals	Manure	Atmos. Depos.	Seeds	Bedding	Other	Animals	Animal products	Crop offtake	manure	Other
Netherlands	National	Van Eerdt and Fong, 1998	x	x	x			x	x		Stock changes, sewage sludge and urban compost	x	x	x	x	Feed
	6 farms. Adapted from MINAS	Van Beek et al., 2003	x	x		x	x					x	x		x	
New York state	Case study farm. Closer to a whole farm budget	Hutson et al., 1998	x	x	x	x	x				Internal cycling of crops and excreta	x	x	x	x	NH3, NO2, NO3
New Zealand	4 farmlets	Ledgard et al., 1999	x	x	x			x					x	x		
	'Typical' New Zealand farm. OVERSEER	Ledgard et al., 2004	x	x	x		x	x			Irrigation	x	x	x		
Scotland	Catchment	Domburg et al., 2000	x	x	x	x		x	x			x	x	x		
	2 farmlets	Watson and Atkinson, 1999	x	x		x			x	x		x		x		
Sweden	138 farms	Swensson, 2003	x		x		x	x	x			x	x	x		
Switzerland	National	Herzog et al., 2008	x	x	x		x	x	x			x	x			
UK	Environmental management for agriculture	Goodlass et al., 2003	x	x		x	x				Irrigation, soil analysis, soil N supply	x		x	x	Gaseous losses
	National	Lord et al., 2002	x	x	x						Sewage sludge	x	x	x		
	Demonstration dairy farm	MIDAS – ADAS (1999)	x	x		x		x				x	x	x		
	171 farms nationwide	PLANET – Defra, 2005	x	x	x	x	x			x		x	x	x	x	

Table A-3: Exploration of mitigation methods for mitigation scenarios

Category	Mitigation method	Nutrient budget sensitivity	Applicability to farm scale analysis	Simulation approach	Mitigation scenario (see section 6.4.1)
Land use	Convert arable land to extensive grassland	Reduction in fertiliser input and crop output.	Field scale surplus would effectively become zero therefore better suited applicable to the farm scale.	Re-calculate with % arable area treated as 0 input un-grazed grass.	Convert 20% arable area to grassland (based on 14% uptake of premium grassland option in Nitrate Sensitive Area Scheme (Lord et al., 1999).
Soil management	Establish cover crops in the autumn	Possible reduction in fertiliser **in following year** / increased crop output where removed.	Best suited to field scale evaluation. Not all fields can accommodate cover crops so results diluted at farm scale.	Reduce fertiliser input by crop available N from cover crops on all previously cover cropped fields.	n/a
	Establish in-field grass buffer strips	Reduction in inputs / outputs to / from buffer strip area	Location of field is key to effectiveness (for P and sediment) and its implementation governed by field scale features, however reductions in inputs / outputs will impact on both field and farm results.	Reduce inputs / outputs according to the area occupied by buffer strip – assumed to be 10% of field area in Cuttle et al., (2006).	n/a
Livestock management	Reduce overall stocking rates on livestock farms	Adjustment of livestock related inputs and outputs	Farm scale simulation preferable to avoid issues related to distribution of grazing and excreta/ urine, and time spent in housing.	Reduce livestock numbers and related inputs and outputs by a proportional amount.	In agreement with Fezzi et al., (2008) and Cuttle et al., (2004) livestock density reduced by 20%. Livestock related inputs and outputs adjusted by a proportional amount.
	Reduce the length of the grazing day or grazing season	Increase in feed, reduction in fertiliser; however changes may balance. Greater effect on distribution of N rather than total inputs.	Applicable at both scales. Farm scale investigations would allow consideration of compensatory increase in feed imports; however simulation would be more complex and uncertain.	Reduce grazed grass fertiliser. Calculate required increase in feed.	n/a

Category	Mitigation method	Nutrient budget sensitivity	Applicability to farm scale analysis	Simulation approach	Mitigation scenario (see section 6.4.1)
	Reduce dietary N and P intakes	Reduction in N content of feed / quantities imported. Change in cropping (increase maize)? Possible reduction in manure or manure N.	Reduce feed import and increase area of maize – both farm scale factors.	Reduce feed N or feed quantity. Adjust maize area / manure situation.	Reduce feed CP to 14% in line with Cuttle et al., (2006)
	Adopt phase feeding of livestock	Feed input / manure output	Farm scale only	Difficult to simulate as require information regarding the distribution of feed amount livestock. Changes in feed N may be balanced by changes in manure N	n/a
Fertiliser management	Use a fertiliser recommendation system	Reduced fertiliser imports, increased manure out	Applicable at field and farm.	Reduce fertiliser to recommended levels.	Calculate budget using recommended fertiliser applications for 2008 crops.
	Integrate fertiliser and manure nutrient supply	Reduced fertiliser in, increased manure out	Most applicable at farm scale if total excreta considered.	Subtract total farm crop available N from fertiliser.	Calculate total farm readily available N and subtract from total farm fertiliser.
	Reduce fertiliser application rates	Reduced fertiliser in, reduced manure out	Applicable at field and farm but more directly linked to field budgets.	Reduce fertiliser applications by 20%	n/a
Manure management	Increase the capacity of farm manure (slurry) stores	Affects N content of manure but no sensitivity to adjustments in timing of application.	Effect on N content similar on all fields. Timing would be field specific.	Sensitivity too low	n/a
	Minimise the volume of dirty water produced	Affects manure export / reduced N content.	Applicable at farm and field scale	Self regulating?	n/a
	Adopt batch storage of solid manure	Reduced N content.	Applicable at farm and field scale	Reduced N content	n/a
	Compost solid manure	Reduced N content	Applicable at farm and field scale	Reduced N content	n/a

Category	Mitigation method	Nutrient budget sensitivity	Applicability to farm scale analysis	Simulation approach	Mitigation scenario (see section 6.4.1)
	Change from slurry to a solid manure handling system	Type of manure exported would change. Low sensitivity to adjustments in application timing and availability of N.	Applicable at farm and field scale	Self regulating?	n/a
	Transport manure to neighbouring farms	Manure export	Most applicable at farm scale	Fully account for manure in fertiliser requirements and export any remaining manure. Similar to integration of manure in fertiliser supply.	n/a
	Incinerate poultry litter	Manure export	Applicable at both field and farm, but more intuitive to apply at farm scale.	Export all poultry manure where produced. Low applicability in MSA / EMEL	n/a

Table A4: Examples of farmgate surpluses (kg N ha^{-1}) available in the literature from which values in Table 6-11 were derived. Figures in blue denote values less than the MSA/EMEL upper quartile. Figures in red denote those that exceed the upper quartile.

Farm type / MSA+EMEL surplus (kg N ha^{-1})	Reference	Origin	Details – needed?	Surplus (kg N ha^{-1})
Cattle and sheep	Jurgen et al., 2006	Germany		81-120 / -18-15[a]
65.7	Defra, 2005	UK	171 farms - not representative	186
	Verbruggen et al., 2005	Belgium	National study	195
	Domburg et al., 2000	Scotland	Catchment study	118[b]
	Bassanino et al., 2007	Italy	41 farms	257 / 100[c]
	Watson and Atkinson, 1999	Scotland	Farmlet investigation	285 / 17[d]
Cereal	Defra, 2005	UK	171 farms - no rep all farm types	66 / 95[e]
60.7	Domburg et al., 2000	Scotland	Catchment	70
	Loges et al., 2009	Germany	32 farms	72.9
	Ondersteijn et al., 2002	Netherlands	194 farms rep of national	77
	Langeveld et al., 2005	Netherlands	Commercial arable farms	80-90 / >160[f]
	Dalgaard et al., 2002	Denmark	41 farms	87
	Verbruggen et al., 2005	Belgium	National	139
Dairy	Loges et al., 2009	Germany	32 farms	117
215.2	Ledgard et al., 2004	New Zealand	Average NZ farm	135
	Domburg et al., 2000	Scotland	Catchment	173
	Nielsena and Kristensen, 2005	Denmark	75 progressive farms	175
	Swensson, 2003	Sweden	138 dairy farms	180 / 157[g]
	ADAS (1999)	UK	Demonstration farm	226
	Nevens et al., 2006	Belgium	120 farm rep of country	238
	Defra, 2005	UK	171 farms - no rep all farm types	248
	Verbruggen et al., 2005	Belgium	National	262
	Dalgaard et al., 2002	Denmark	41 farms	289
	Ondersteijn et al., 2002	Netherlands	194 farms rep of national	313
	Bassanino et al., 2007	Italy	41 farms	318
Mixed	Defra, 2005	UK	171 farms - not rep of farm types	76 / 136 / 152 / 248[h]
97.7	Domburg et al., 2000	Scotland	Catchment	119
	Dalgaard et al., 2002	Denmark	41 farms	197
	Ondersteijn et al., 2002	Netherlands	194 farms rep of national	231 / 285[i]

[a] Intensive / extensive farms
[b] Lowland cattle and sheep
[c] Beef / suckler farms
[d] Fertiliser / N fixation reliant
[e] Farm without / with imported manure
[f] Sand / clay

[g] Intensive / extensive farms
[h] No crop export / crop export
[i] Arable and C+S / arable and dairy / arable and mixed livestock / arable and pigs and poultry
[j] Arable and intensive livestock / arable and dairy

Table A-5: Fertiliser N uptake and use efficiencies (%) in the literature. N uptake efficiency refers to the proportion of applied N captured by crops, inclusive of N in roots and stems. N use efficiency refers to the proportion of applied N harvested in grain. Values in Sylvester-Bradley and Kindred (2009) were favoured due to their UK origin and wide range of crop for which data was available. Values exceed 100% due to soil N supply.

Crop	'All' literature						Sylvester-Bradley and Kindred (2009)	
	N uptake efficiency (NupE) (%)			N use efficiency (NUE) (%)			NupE	NUE[a]
	Mean	*Max*	*Min*	Mean	*Max*	*Min*		
Grass	80	90	70	80	90	70	-	-
Linseed	145	145	145	117	117	117	145	117
Maize	58	58	58	-	-	-	-	-
Rye	98	148	48	103	103	103	148	103
Spring barley	76	108	56	70	84	55	108[b] / 65[c]	84[b] / 55[c]
Spring wheat	81	109	53	83	83	83	109[d]	83[d]
Spring oats	106	106	106	73	73	73	106	73
Spring oil seed rape	176	176	176	56	56	56	176	56
Triticale	107	148	67	109	109	109	148	109
Winter barley	68	96	55	40	65	28	96[b] / 70[c]	65[b] / 49[c]
Winter wheat	66	93	43	41	72	31	93[b] / 90[d]	72[b] / 67[d]
Winter field beans	65	65	65	51	51	51	65	51
Winter oats	116	116	116	83	83	83	116	83
Winter oilseed rape	118	118	118	37	46	26	118	46
Average [e]	**99**			**72**			**110**	**74**

[a] Derived from average N applied in Sylvester-Bradley and Kindred (2009) divided by N offtake in Sylvester-Bradley (1993). N uptake comparable between the two studies therefore N offtake assumed to be similar.

[b] feed varieties

[c] malting varieties

[d] milling varieties

[e] Average of mean values to ensure equal weighting of each crop

Table A-6: Examples of farm type efficiency (%) presented on a farm type basis found in the literature. Average values presented in Table 7-13.

Farm type	Reference	Country	Additional details	Value
Cattle and sheep	Domburg et al., 2000	Scotland	Lowland cattle and sheep farms	38
	Bassanino et al., 2007	Italy	Beef breeding farms	26
	Bassanino et al., 2007	Italy	Suckling cow farms	19
	Verbruggen et al., 2005	Belgium	Beef	33
	Average			**29**
	Max.			**38**
	Min.			**19**
Cereal	Aronsson et al., 2007	Sweden	Clay soils	54
	Domburg et al., 2000	Scotland		73
	Ondersteijn et al., 2002	NL		51
	Verbruggen et al., 2005	Belgium		68
	Average			**62**
	Max.			**73**
	Min.			**51**
Dairy	Aarts et al., 2000	NL	Typical Dutch commercial farm	14
	Aarts et al., 2000	NL	de MArke demonstration farm	31
	Bassanino et al., 2007	Italy		34
	Domburg et al., 2000	Scotland		28
	Hristov et al., 2006	US	Maximum of 64% where satisfying own forage needs and high yielding cattle. Would increase to 68% if NH_3 loss accounted for.	41
	Leach and Bax, 1999	Scotland	Intensive grassland dairy in Scotland including N fixation and atmospheric deposition under experimental conditions	18
	Ledgard et al., 1997	UK	Inclusive of N fixation	30
	Ledgard et al., 1997	UK	Inclusive of N fixation and atmospheric deposition	20
	Ledgard et al., 1999	NZ	Farmlets receiving 400N ha^{-1}	28
	Ledgard et al., 1999	NZ	Farmlets receiving 0N	52
	Mabon et al., 2009	France	Grassland system	29
	Mabon et al., 2009	France	Dairy plus feed crop production	40
	Mabon et al., 2009	France		27
	Nevens et al., 2006	Belgium	Inclusive of N fixation and atmospheric deposition	19
	Nielsen and Kristensen, 2005	Denmark		24
	Ondersteijn et al., 2002	NL	via MINAS calc	24
	Powell et al., 2009	n/a	Biological maximum efficiency	35
	Roberts et al., 2007	Scotland		17
	Swensson, 2003	Sweden	Ave 21% where no crop output, Ave 29% where crop export	27
	Treacy et al., 2008	Ireland		19
	Van Bruchem et al., 1999	NL	De Ossekampen demonstration farm on heavy clay	29
	Van Wepern, 2009	NL	Increased to 39 following advice and mitigation	30
	Verbruggen et al., 2005	Belgium		22
	Hristov et al., 2006	US		41
	Powell et al., 2006	US		25
	Average			**28**
	Max.			**52**
	Min.			**14**

Farm type	Reference	Country	Additional details	Value
Mixed	Domburg et al., 2000	Scotland		37
	Van Bruchem et al., 1999	NL	APMingerhoudhoeve demonstration farm on loamy soil	64
	Ondersteijn et al., 2002	NL	Arable and intensive livestock	66
	Ondersteijn et al., 2002	NL	Arable and dairy	35
	Average			**51**
	Max.			**66**
	Min.			**35**

Table A7: Time and cost associated with budget and measurement assessment methods. Worked example for MSA. All prices and times approximate.

<table>
<tr><th rowspan="2">Method</th><th colspan="4">Resources (time / cost)</th><th rowspan="2">Timescales</th><th rowspan="2">Specialist equipment</th><th rowspan="2">Who??</th></tr>
<tr><th>No. samples</th><th>Frequency / duration</th><th>Sampling time</th><th>Analysis</th></tr>
<tr><td rowspan="3">SMN</td><td>28 fields, 1 sample per field</td><td>Twice a year (late autumn / early winter plus late winter / early spring)</td><td>Using Hydrocare 0.75hrs per field inc travel time plus 1hr prep per day (machine and sampling plans / bags) and 1 hr analysis prep. 2 people required for health and safely. Crop N sampling (spring sample only) = 0.2hrs per field but carried out by second person

Total sampling time = 28 x 0.75hrs = 21hrs
Assuming 6hr available after prep = 4 working days required</td><td>1 soil sample per field @ £11
X 28 fields
X 2 sampling occasion plus 1 crop N sample per field @ £6 x 28 fields x1 sampling occasion</td><td rowspan="3">4 years +?? Check time scales of results in PP / SMN results table</td><td rowspan="3">Hydrocare
Vehicle with tow bar</td><td rowspan="3">Farmer – details of soil N facilitates more accurate fertiliser applications
Consultants / contractors / scientists</td></tr>
<tr><td colspan="3">Total time = 4 days x 2 sampling occasions x 2 people = <u>128 hrs</u>
<u>Total labour cost = 128hrs x £10 = £1280</u></td><td>Total analysis cost = (1 x £11 x 28 x 2) + (1 x £6 x 28 x 1) = <u>£784</u></td></tr>
<tr><td colspan="4">OVERALL COST = £2064</td></tr>
<tr><td rowspan="3">PP</td><td>27 fields, 5 samples per field</td><td>Fortnightly samples from Nov – Apr = approx 12 sampling occasions</td><td>Installation = approx. 1hr per field. 2 people required for health and safely, plus 1 hour prep per day.
Sampling = approx. 0.5hrs per field inc. travel time plus one hr prep per day

Total installation time = 27 x 1hr = 27hrs
Assuming 7hrs available after prep, 4 working days required

Total sampling time = 27 x 0.5 = 13.5hrs
Assuming 7hrs available after prep, 2 working days required</td><td>5 samples per field @ £2.50
X 27 fields in catchment x 12 sampling occasions</td><td rowspan="2">As above</td><td rowspan="2">Hydrocare
Vehicle with tow bar</td><td rowspan="2">Consultants / contractors / scientists</td></tr>
<tr><td colspan="3">Total time = (4 days installation x 2 people) + 12 x 2 days sampling = 28 days = 28 x 8hrs = <u>288 hrs</u>
<u>Total labour cost =288 x £10 = £2880</u></td><td>Total analysis cost = 5 x £2.50 x 27 x 12 = <u>£4050</u></td></tr>
<tr><td colspan="4">OVERALL COST = £6930</td><td></td><td></td><td></td></tr>
</table>

Method	Resources (time / cost)				Timescales	Specialist equipment	Who??
	No. samples	Frequency / duration	Sampling time	Analysis			
WQ measurement	72 sites, 1 sample per site	Monthly	Sampling = 0.25hrs per site Prep = 0.5hrs per day Total sampling time = 72 x 0.25 = 18hrs Assuming 7.5hrs available after prep, 3 working days required	1 sample per site x 72 x £2.50 x 12 sampling occasions	Long term	Pump for borehole samples	Environment agency
	Total time = 12 x 3 days sampling = 36 days = 36 x 8hrs = 228 **<u>hrs</u>** **<u>Total labour cost =228 x £10 = £2280</u>**			Total analysis cost = 72 x £2.50 x 12 = £2160			
	OVERALL COST = £4440						

Method	Resources (time / cost)			Timescales	Specialist equipment	Who??
	No. samples	Frequency / duration	Calculation time			
Soil surface budgets	Xxx fields on 18 farms, 1 budget per field	Annual	Data collection = 1 day per farm = 18 days (note that collected data from more than 1 year) Data manipulation and calculation = 1 day per farm = 18 days Total calculation time = 18 + 18 days = 36 days			Farmers, scientists
	Total time = 36 days = 36 x 8hrs = 288 **<u>hrs</u>** **<u>Total labour cost =288 x £10 = £2880</u>**					
Farmgate budget	18 farms	Annual	Data collection = 0.5day per farm = 9 days (note that data collected from more than 1 year and can also be used to calculate efficiency) Data manipulation and calculation = 0.5 days per farm = 9 days Total calculation time = 9 + 9 days = 18 days			Farmers Catchment advisers
	Total time = 36 days = 18 x 8hrs = 144 **<u>hrs</u>** **<u>Total labour cost =144 x £10 = £1440</u>**					
Efficiency	18 farms	Annual	Data collection = 1day per farm = 18 days (note that data collected from more than 1 year and can also be used to calculate efficiency) Data manipulation and calculation = 1 days per farm = 18days (note that calculation time would be less if total efficiency was calculated only) Total calculation time = 18 + 18 days = 36 days			Farmer, catchment advisers
	Total time = 36 days = 36 x 8hrs = 288 **<u>hrs</u>** **<u>Total labour cost =288 x £10 = £2880</u>**					