



Sustainable nutrient management at field, farm and regional level: Soil testing, nutrient budgets and the trade-off between lime application and greenhouse gas emissions



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ABSTRACT

Pollution from agriculture has environmental consequences at local and global scales. Managing this pollution is challenging because of diffuse sources and complex relationships between aquatic and atmospheric emissions. We illustrate this for a UK county that has suffered outbreaks of microbial pollution and eutrophication. We surveyed 49 livestock farms covering 12% of total agricultural grazed land. Soil nutrient status and whole-farm nutrient balances were determined, and the environmental impact of alleviating sub-optimal soil pH by liming was estimated at the county level. Only 37% of fields contained more P than was required for satisfactory grass growth, and soil acidity and available K were often limiting production. The mean farm N, P and K balances were similar to a modelled farm in England & Wales and EU indicators for the majority of North West Europe. This suggests that local eutrophication events linked to agriculture are more likely to relate to improper timing of nutrient application rather than over-application. None of the surveyed farmers used nutrient decision support tools, largely due to a lack of awareness and competing sources of information. Liming soils to pH 6.0 was estimated to both reduce N-leaching and N₂O emissions; however, the net climate-change impact would be negative as the direct CO₂ emissions would exceed CO₂ equivalent emissions of not liming by 394% (95% CI 201–21,232). Although liming currently presents a net cost to farmers, a sensitivity analysis suggests that reduced lime cost could lead to economic benefit to farmers but still increased greenhouse gas emissions. The results are applicable to all pasture-based agricultural systems where there is a drive to maintain or increase production through optimal soil and nutrient management. The findings demonstrate an important trade-off between reducing aquatic and atmospheric pollution and agricultural productivity, and the need to improve communication of this trade-off to farmers.

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1. Introduction

Maintaining agricultural production, while minimising diffuse pollution to water and air, is a global problem. Direct emissions from agriculture comprise roughly 11% of global greenhouse gas emissions and these emissions are projected to rise by 20% by 2030 (US-EPA, 2011). Including indirect emissions increases the total emissions from agriculture to 19–29% of the global total (Vermeulen et al., 2012). Anthropogenic activities have profoundly altered the global nitrogen and phosphorus cycles and will continue to do so (Bouwman et al., 2009). Net anthropogenic nitrogen

inputs in China, US & Northern Europe are estimated at between 2 and 3.5 t ha⁻¹ of which 15–30% is exported in rivers (Swaney et al., 2012). Indeed, studies across the globe have shown agriculture to be amongst the largest contributor of annual nitrate and phosphate loads to river waters (Defra, 2007; Puckett et al., 2011; Liu et al., 2012).

To advance sustainable agricultural production requires considering the management activities at the scale of farm and field, in the context of policy at the regional or catchment level. In this paper, we consider the management of agricultural inputs and the consequent impact on diffuse pollution at three different scales: field, farm and region. Common field and farm management activities affecting diffuse pollution include the over-application of fertilizer (Lord and Mitchell, 1998; Withers et al., 2001), the inappropriate application of manure or slurry to land (Smith et al., 1998; Shepherd

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et al., 2001; Shepherd and Chambers, 2007), or poor management of soil leading to erosion and surface runoff on both livestock and arable farms (Quinton et al., 2010). However, although such activities are often very localized and spatially explicit, monitoring programmes and policy to reduce emissions are largely applied at the catchment and regional scale. For example, regulations such as the Water Framework Directive, which requires all surface waters in the European Union (EU) to be of 'good ecological status' by 2015 (WFD, 2000). Also in the EU, policy-makers have endeavoured to reward farmers for reducing the risk of diffuse pollution such as by managing land under 'Good Agricultural and Environmental Condition' to benefit from the Single Payment Scheme. The European Commission has adopted 28 indicators of agri-environment status to assess the interaction between the Common Agricultural Policy and the environment at EU, national and regional level. A subset of these indicators are directly related to the risks and impacts of diffuse emissions to water and air viz. nitrogen and phosphorus surpluses, greenhouse gas emissions and nitrate leaching.

While indicators provide guidance on levels of emissions, effective management of diffuse pollution requires both that farmers are able to connect their management activity with diffuse pollution outcomes and that there are incentives to adopt sustainable management. A frequently used approach to quantify whether a farm is managing nutrients efficiently, and identify ways to improve nutrient-use efficiency, is to construct a nutrient budget (e.g. Koelsch, 2005; Kettering et al., 2012; Langeveld et al., 2007; Cherry et al., 2012) which is a powerful tool for raising awareness and stimulating action (Goulding et al., 2008). Voluntary best management practice supported by nutrient budgets have been shown to be more effective in reducing diffuse pollution than regulation (Koelsch, 2005). On livestock farms a 'farm-gate' nutrient budget (e.g. PLANET; Defra, 2005) is generally considered the most versatile method of doing this, which takes into account nutrient loads coming in and going out of the farm-gate. Further, a farm-gate budget is deemed to be a suitable environmental performance indicator (Oenema et al., 2003). Whilst uncertainties exist in all budget methodologies, they are usually smaller for a farm-gate budget and these are therefore preferred over, e.g., soil surface budgets, as a policy instrument (Oenema et al., 2003). The result, usually a surplus of N or P, is compared against calculated benchmarks. The surplus nutrient is thus used to indicate the relative risk of diffuse pollution (Lord et al., 2002; Oborn et al., 2003) and can be used to derive farm-level nutrient-use efficiency. Nutrient budgeting has been employed by policy-makers (e.g. PARCOM, 1988; EEA, 2001; OECD, 2001) as an agri-environmental indicator and to raise awareness of nutrient use efficiency. Levels of N and P surplus are two indicators in the EU set of 28 and they have also been used as a regulatory policy instrument, e.g. in the Netherlands (MINAS) where it has been shown that by reducing whole-farm surpluses, surface and groundwater concentrations of N and P decline over time (Oenema et al., 1998). Given the current high cost of fertilizers, nutrient budgeting can also be valuable in highlighting to farmers potential imbalances and, when combined with standard field soil testing (for pH, P, K and Mg), quantify incentives for reducing inputs. However, uptake by farmers of decision support systems in general has been poor (Matthews et al., 2008; Hochman and Carberry, 2011) and there seems to be little evidence that nutrient management is an exception.

Some nutrient budgeting programs (e.g. PLANET; Defra, 2005) do not consider soil pH, although it is well documented that sub-optimal levels of pH (<6.0) prevent optimal plant nutrient-use efficiency and can increase surplus soil N (Stevens and Laughlin, 1996) which could exacerbate N leaching. The mechanism for increased growth and N uptake is complex, arising both from the alleviation of toxicity (from Al and Mn) and the alleviation of nutrient deficiency (of Ca, Mg and Mo). Liming is the most

common method of raising the pH of acidic soil (Goulding et al., 1989; Viade et al., 2011); however, recent data (British Survey of Fertiliser Practice, 2010; AIC, 2011) show that there has been a considerable decrease (20–30%) in lime applications to agricultural grassland in the UK since the mid-1990s. Similarly, in Ireland the amount of lime used on grassland has dropped greatly in recent years (Tunney et al., 2009). This reduction in the use of lime, together with the acidifying effects of high rates of N fertilizer application, may therefore be limiting crop production and exacerbating N losses. Whilst the limited existing research indicates that lime extraction and application to agricultural soils can lead to greenhouse gas (GHG) emissions (IPCC, 2006), it is well known that fertilizer production and application also increases GHG emissions (IPCC, 2006), as well as the loss of nutrients to surface and ground waters described previously. The application of lime may enable farmers to reduce fertilizer application rates whilst still maintaining crop productivity due to better utilization of the nutrients applied and stimulation of nutrient cycling from native soil organic matter reserves. This may have multiple benefits in reducing both on-farm costs and nutrient losses through leaching and GHG emissions. Such information is needed by industry and policy-makers to stimulate 'win-win' management practices that deliver truly sustainable farm systems (Goulding et al., 2008).

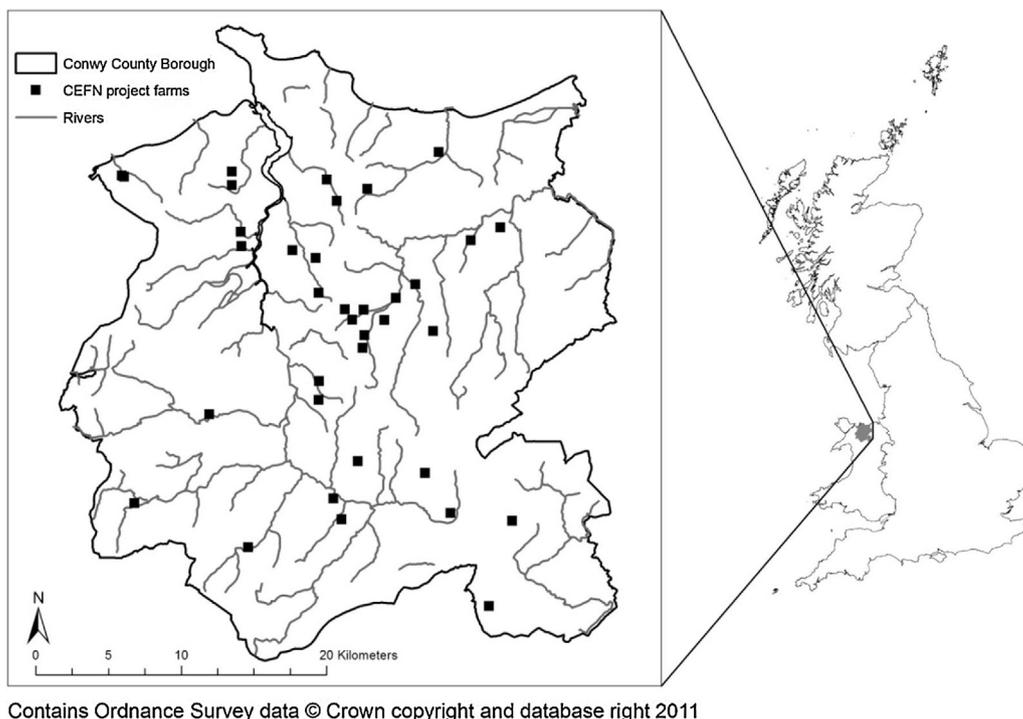
The aims of this study were to explore sustainable nutrient management at the field, farm and regional level for one rural UK county in terms of: (i) variation amongst farms in field level nutrients; (ii) whether current levels of surplus N, P and K are likely to lead to on-farm economic losses and risk of environmental pollution; (iii) the regional-level environmental trade-off (nutrient leaching versus GHG emissions) and economic costs of reducing nutrient application through increasing lime application. The work was done as part of the 'CEFN Conwy' project, which aimed to reduce diffuse nutrient and microbial pollution from agriculture by assisting farmers in the sustainable on-farm management of nutrients and soils (CEFN Conwy, 2011).

2. Materials and methods

2.1. Study area and farmer recruitment

The fieldwork was carried out during November 2009 and February 2010 within the county of Conwy in north-west Wales, UK (Fig. 1). The county's area is 1130 km² (Conwy Council, 2012), and is typical of many parts of the UK, with land types ranging from relatively unproductive waterlogged mineral soils and organic peat soil moorland in upland areas, which predominantly support extensive sheep farming and commercial forestry, to fertile lowland areas that support a variety of livestock farming. Its drainage is dominated by two large river systems fed by a number of smaller catchments and sub-catchments (Fig. 1). In general, Conwy has very few water bodies designated as 'poor' in water quality status relative to Water Framework Directive targets (EA, 2009), but several are classified as 'moderate' and are potentially at risk from diffuse pollution. The region is not currently in a Nitrate Vulnerable Zone (in such regions farmers must follow restrictions designed to reduce nitrate losses). The estuary area of the River Conwy is important for the commercial harvesting of shellfish and has several public beaches with designated EU bathing waters, and these have in the past been affected by contamination with faecal coliforms (Thorn et al., 2012).

Farmers across the county were recruited by a combination of awareness-raising events and local press, and through the two largest farmers' unions. Forty-nine farmers from across the county agreed to undertake a farm survey in return for free soil testing of at least two fields of their choice. Farm types were representative



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Fig. 1. Location of Conwy County in the UK and sampled farms within the county. The main river systems of the county are shown. Total area is 1130 km² (Conwy Council, 2012).

of those within the catchment (mixed beef/suckler cow and sheep>sheep only>dairy only), mostly on improved grassland.

2.2. Soil survey

Surveyed farmers selected their improved fields (i.e. excluding rough grazing) for soil sampling; they chose their ‘best’ and ‘worst’ fields (in terms of livestock production), to capture the range of soil nutrient status. Fields were sampled using a ‘W’ grid of 25 cores at 7.5 cm taken over the whole field, equally spaced, and a composite sample produced (Defra, 2010). Samples were boxed and sent to a nationally accredited laboratory specialising in agri-soil analysis (NRM Ltd., Berkshire, UK), where soil pH, P (Olsen’s P extraction), potassium (K) and magnesium (Mg) (1 M ammonium nitrate extraction) were measured. In the UK, for fertilizer recommendation purposes, soil analysis results for P, K and Mg are reported as an Index, where Index 0 represents very low fertility (i.e. deficient), Index 1 is low, Index 2 is adequate (for most crops) and Index 3 and above indicates unnecessarily high fertility for grassland and arable crops (Defra, 2010). (For concentrations by index see Table 1.) The results were compared with two national datasets covering all types of grassland: the Representative Soil Sampling Scheme (RSSS) results for 1995–99 (1454 fields) in England and Wales (the last cycle of data before the scheme was abolished in 2001) and the UK-wide data collated by the Professional Agricultural Analysis Group (PAAG) in 2009/10 (ca. 200,000 fields), the same year as the Conwy survey. Whereas the RSSS dataset represents a fully randomized sample (see Skinner and Todd, 1998), the PAAG data are biased towards those more technically aware farmers who send in soil samples to commercial laboratories (PAAG, 2011). Providing both datasets gives a comprehensive picture of farmer’s fields.

To test how representative the sample fields were of the improved grassland in Conwy county, polygons representing the sample fields were obtained from the OS MasterMap Topo_Area

spatial data component (Ordnance Survey, 2010) via the EDINA Digimap service. The population of improved grassland fields within the county were selected using the topography layer land theme ‘natural feature’ class, where the polygon centroids intersected the ‘improved grassland’ land cover class contained within the Countryside Council for Wales phase 1 habitat classification vector dataset (JNCC, 2007; Howe et al., 2005). The resulting 46,117 polygons were then used together with OS Landform Profile Digital Terrain Model (10 m grid) within ArcGis 9.3.1 (ESRI 1999–2009) to calculate zonal statistics (mean field elevation, mean field slope, mean field aspect).

2.3. Farm nutrient budgeting

2.3.1. Data collection

Surveys were undertaken by interviewing the farmers about the import and export of all goods, manures and livestock during the year 2009. Farmers were also asked a range of socio-economic questions covering such subjects as their decision-making processes, where they sourced information on soil and nutrient management, how important they viewed the issue, etc. Farm-gate nutrient balances were estimated using the same methodology as PLANET nutrient management software (Defra, 2005). N, P and K nutrient surpluses and deficits were calculated as the difference between nutrients imported (in feed, livestock, bedding, mineral fertilizer, organic manures and wastes, legume-fixed N and deposited N) and exported (in livestock and livestock products, crops and manures). It was assumed that in improved permanent pasture, clover fixed 40 kg N ha⁻¹ y⁻¹ (an approximation based on Kristensen et al. (1995) and Halberg et al. (1995)), and also that the mean N wet deposition for Conwy was 21 kg N ha⁻¹ y⁻¹ (Defra, 2011). Of the 49 farms surveyed, 36 provided sufficient information for a nutrient balance to be calculated. Farm nutrient balances were compared with published best-practice balances for beef/suckler cow farms (the closest match for available data) (Defra, 2005), and European

Table 1

Distribution percentages of soil fertility analyses from 99 fields from 48 farms in Conwy County, Wales. Comparative national data from the Representative Soil Sampling Scheme (RSSS) for 1995–99 for England and Wales and from the Professional Agricultural Analysis Group (PAAG) for 2009/10 for the UK are also given (Skinner and Todd, 1998; PAAG, 2011). The Range column gives the soil indices concentrations in mg l⁻¹ for P (Olsen's method), K (ammonium nitrate extract) and Mg (ammonium nitrate extract).

Index	P				K				Mg			
	Range	Conwy	RSSS	PAAG	Range	Conwy	RSSS	PAAG	Range	Conwy	RSSS	PAAG
0	0–9	1	11	9	0–60	5	5	8	0–25	0	0	0
1	10–15	14	28	19	61–20	54	39	37	26–50	2	3	4
2	16–25	48	31	29	121–240	36	27	40	51–100	35	34	31
3	26–45	33	22	31	241–400	5	17	12	101–175	60	34	38
4+	>45	4	8	12	>400	0	12	3	>176	3	29	27

RSSS is an unbiased structured survey that captures the full range of grassland systems and management and is therefore representative of a range of grassland management practices. PAAG results are based on samples received by a group of commercial laboratories and will be biased towards those farmers who value soil analysis as a management tool.

Union estimated gross nitrogen and phosphorus balances (Eurostat, 2011).

2.3.2. Calculating level of N surplus at catchment level

Total annual N surplus from farmed grassland was estimated using:

$$N_C = G \cdot N_F \quad (1)$$

where N_C is total grassland N surplus (in kg), G is area (ha) of farmed grassland in the county of Conwy (EDINA, 2011), and N_F is mean N surplus of the surveyed farms, weighted by farm area (kg N ha⁻¹).

N_C was partitioned into leached-N and demineralized-N using summary figures from Watson and Atkinson (1999) for grass-clover swards. The concentration of leached N from the root zone was then estimated as:

$$N_{\text{conc}} = \frac{1000 \cdot N_{\text{Cl}}}{G \cdot R} \quad (2)$$

where N_{conc} is concentration of leached N from farmed grassland (as mg N l⁻¹), N_{Cl} is the annual quantity of leached N (in kg), and R is mean flow for the Conwy catchment (in mm, equivalent to m³ ha⁻¹; Littlewood, 2003). Direct N losses as nitrous oxide (N₂O) were estimated using the results of Cardenas et al. (2010) (parameters in Table 4c of Cardenas et al. (2010) for the Rowden and Cae Banadl sites) and the mean annual N inputs of the surveyed farms:

$$N_{\text{N}_2\text{O}} = \frac{-4330 + 5062 \cdot 1.00564^{N_{\text{in}}}}{1000} \quad (3)$$

where $N_{\text{N}_2\text{O}}$ is N-N₂O emissions (in kg N ha⁻¹ y⁻¹) and N_{in} is the level of N inputs (in kg N ha⁻¹ y⁻¹). The goal of the Cardenas et al. (2010) research was to refine the IPCC values for N₂O for UK conditions, so it is more robust than the generic IPCC constants. Indirect N₂O losses from leached N were calculated using standard IPCC methodology (IPCC, 2006).

2.4. Calculating N offset through lime application

Using the field-testing results, mean farmed grassland pH and total lime requirements were estimated from farm means, weighted by amount of improved pasture on each farm, and assuming a five-year gap between liming events (Chambers and Garwood, 1998). Following national RB209 guidelines (Defra, 2010), it was assumed that soils were limed to an optimum pH of 6.0 and that lime was only applied to improved grassland. The reduction in N surplus produced by liming to recommended levels was taken from the experimental results (increase in N uptake for limed grassland with 80 kg ha⁻¹ N fertilizer input) of Stevens and Laughlin (1996).

The CO₂-equivalent greenhouse gas emissions footprint of the lime requirements was estimated using standard IPCC methodology (IPCC, 2006) with the additional embodied C in mining and preparing lime and transport from point of sale to farm gate

included. Estimated total grassland N₂O emissions were converted to CO₂-equivalent using IPCC standard global warming potentials (298; IPCC, 2007).

The uncertainty of the effect of liming on mean N uptake and GHG emissions was explored using 10,000 Monte Carlo draws for farm N surplus (N_F), fertilizer N inputs (N_{in}), partitioning of surplus N and the effect of liming on N uptake. For the N partitioning, a uniform distribution was used with limits $\pm 20\%$ of the Watson and Atkinson (1999) figure. For the other parameters, a normal distribution was used with the farm standard error (for the N_F and N_{in} parameters) and the Stevens and Laughlin (1996) experimental standard error for the effect of lime on N uptake.

2.5. Cost-effectiveness of substituting N application with lime

Reducing N leaching through applying lime has private costs to farmers of lime application and private benefits from reduced N inputs. Costs of N inputs (\$0.83 kg⁻¹) and lime application (\$22.5 t⁻¹) are taken from Nix (2009). There are social costs of GHG emissions from the production and use of lime, and social benefits from avoided emissions due to reduced N production and application. There are several ways to estimate the benefit of reduced GHG emissions. The first is to use estimates of the marginal social cost of the emissions, which gives a measure of the benefit to society of emissions reduction. Estimates vary widely. Tol (2005) reviewed 28 studies finding a median of \$14 tC⁻¹ and a mean of \$93 tC⁻¹. The Stern review (Stern, 2006) used rather higher figures, ranging from \$89 to 310 tC⁻¹ depending on the emissions scenario. Therefore, the more recent DECC (UK Department of Energy and Climate Change) guidance (DECC, 2009) was used. This recommends pricing C based on mitigation cost, and gives a 2020 cost of \$37.5 (range \$21–37.5) for traded C and \$90 (range \$45–115) for non-traded C (all DECC figures are for tCO₂-equivalent and converted from GBP to USD using an exchange rate of 1.5) and these are the assumed prices we use here.

3. Results

3.1. Study area and farmer recruitment

Of the 49 farms surveyed, 36 had sufficient data to estimate farm nutrient budgets and soil testing was carried out for 48. The 49 sampled farms had a total area of 9209 ha, 12% of the grassland area within the county. The 36 farms with complete nutrient budgets covered an area of 3964 ha and had 73% improved grassland. For comparison, the total Conwy County area is 113,000 ha, farm grassland is 74,007 ha, comprising 71% improved grassland and 29% rough grazing (EDINA, 2011). There were sufficient data to calculate stocking rates for 45 farms, of which, 4 had a dairy enterprise, 39 a beef enterprise, 43 a sheep enterprise, 1 a chicken enterprise

Table 2
Summary of mean values of components of nutrient budgets (kg ha⁻¹) for 36 farms in Conwy County where whole-farm nutrient budgets were calculated for the year 2009.

Component	N	P	K
Animal production	-4.6	-1.1	-0.4
Arable crops	-0.1	-0.1	-0.1
Mineral fertilizer, clover and N deposition	98.7	4.3	10.0
Feed	8.4	0.1	3.5
Imported bedding and livestock manure	1.3	0.2	1.9
Total	103.7	3.4	14.9

and 36 a combined beef and sheep enterprise. Mean stocking rate was 1.0 livestock units (LU) ha⁻¹ for total farm area and 1.3 LU ha⁻¹ for improved grassland area. For comparison, of the 36 farms where nutrient budgets were calculated, 1 had a dairy enterprise, 31 a beef enterprise, 36 a sheep enterprise and 31 a combined beef and sheep enterprise. Mean stocking rate for this subsample was 0.9 LU ha⁻¹ for total farm area and 1.2 LU ha⁻¹ for improved grassland area.

3.2. Individual field nutrient status

For the sampled fields, there was no significant difference in distribution of elevation (2 sample Kolmogorov–Smirnov test, $p=0.761$) and aspect ($p=0.336$) in comparison with all improved grassland fields in Conwy County. There was a marginally significant difference in slope ($p=0.049$) with steep sloping fields under-represented in our sample. Analysis of the fertility status of soil samples from a total of 99 fields from 48 farms showed that only 15% of fields were low or deficient (index 0 or 1) in P and 81% of fields had a P index of 2 or 3 (Table 1). Over half of fields (59%) were low or deficient in K but soil Mg fertility was high (only 2% of fields had index 0 or 1). Comparing the results with the RSSS and PAAG national datasets, in Conwy a greater proportion of fields had P Index 2 and 3, and a greater proportion of fields had Mg Index 3, but a greater proportion of fields had a K Index of only 1, than the national averages (Table 1). For P, the mean difference between the ‘best’ and ‘worst’ fields was 7.1 mg l⁻¹ (SE 1.0, median 4.2; range 0.2–25.6), for K the mean difference was 47.6 mg l⁻¹ (SE 7.2, median 31.0; range 0.0–225) and for Mg the mean difference was 29.8 mg l⁻¹ (SE 4.1, median 21.0; range 0.0–138.0).

For the 36 farms where a reliable nutrient surplus could be calculated mean mineral fertilizer inputs were 48.4 kg ha⁻¹ N (range 0.0–120.4), 4.2 kg ha⁻¹ P (0.0–10.7) and 10.0 kg ha⁻¹ K (0.0–29.3). Mean surplus N was 104 kg ha⁻¹ (SE 7.39, median 97; range 26–189; mean weighted by farm area 73.2 kg ha⁻¹; SE weighted by farm area 19.4 kg ha⁻¹), surplus P was 3.4 kg ha⁻¹ (SE 0.5, median 2.8; range -0.3 to 11.3) and surplus K was 14.9 kg ha⁻¹ (SE 1.6, median 12.4; range 0.7–37.3). Substantially the biggest contribution to nutrient surplus was mineral fertilizer, with feed the second most important (Table 2). The overwhelming majority of farms (Fig. 2) had nutrient surpluses less than for a model benchmark farm based on average beef and sheep farms in England and Wales (Defra project ES0124 (Defra, 2005)). However, this may be largely explained by the majority of the surveyed farms being more upland in character (i.e. have lower stocking densities) than the benchmark. In comparison with indicator national averages, the mean surplus N was above the European Union average (50 kg ha⁻¹) for the period 2005–2008 and below the Dutch (210 kg ha⁻¹) and Belgian (119 kg ha⁻¹) but comparable to other North Western European countries, including the UK average (101 kg ha⁻¹). The comparisons for phosphorus surpluses follow a similar pattern. There was no significant correlation between estimated farm nutrient surplus and soil testing data for either P or K (Pearson correlation -0.02 for P, 0.06 for K).

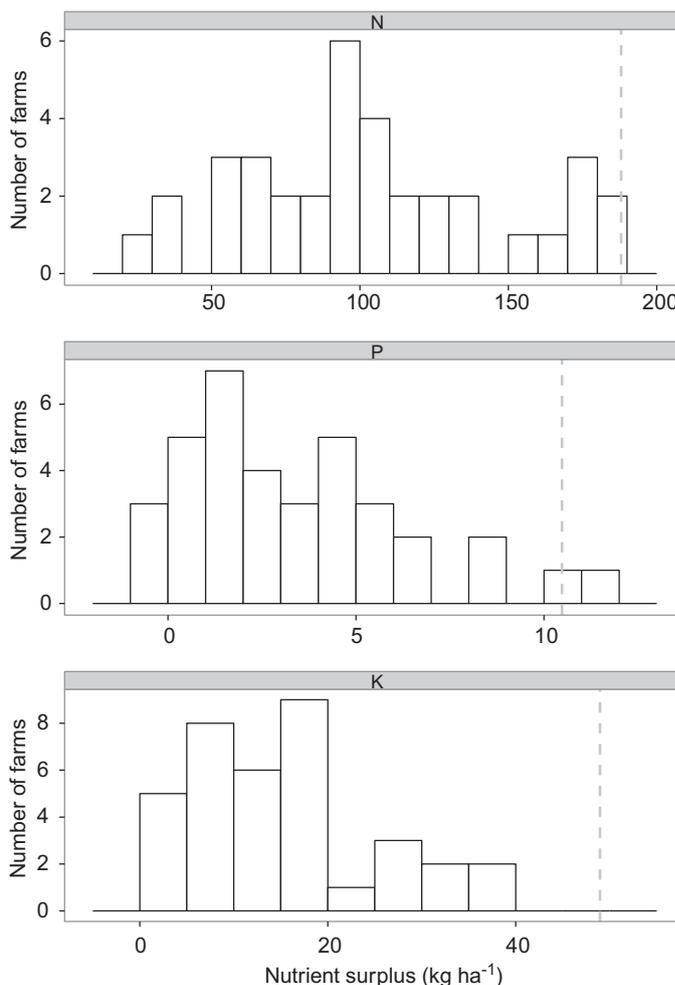


Fig. 2. Distribution of individual farm-level nutrient surpluses for N, P and K estimated using a farm-gate nutrient budget (N = 36) for the year 2009. The dashed grey lines are benchmark surpluses for a beef/suckler cow system (Defra, 2005).

3.3. Soil pH and lime requirements

Of the 36 nutrient budget farms, field nutrient status was available for 35. For the 63 mineral soil (i.e. excluding peat soils) fields on these farms, mean field soil pH, calculated on the anti-log scale, was 5.4 (SE 0.8) with an associated mean lime requirement to reach the recommended pH (6.0) of 3.2 t ha⁻¹. Eighty-eight percent of farms had a mean field pH less than 6.0 (Fig. 3) suggesting that productivity of the whole study area could be increased by liming. Within the surveyed farms, mineral soil fields that had been limed within the last 10 y had a mean pH of 5.7 compared with 5.4 for the non-limed fields (Student’s independent samples t -test $p=0.034$). For these farms the mean difference in pH between the two fields surveyed was 0.29 (SE 0.04, median 0.2; range 0.0–1.1).

3.4. Influences on farmer nutrient-planning

Of the 49 farmers surveyed, none used or had even tried out the two government-sponsored computer programs for nutrient planning (PLANET & MANNER) that are readily and freely available to UK farmers. The most frequent reason given was that they were not aware of these computer programs. Seven farmers had used advice from the UK government’s Environment Agency, 10 farmers the advice of an independent agricultural consultant (ADAS, the UK’s largest independent provider of rural environmental consultancy) and nine farmers obtained advice from other sources, including

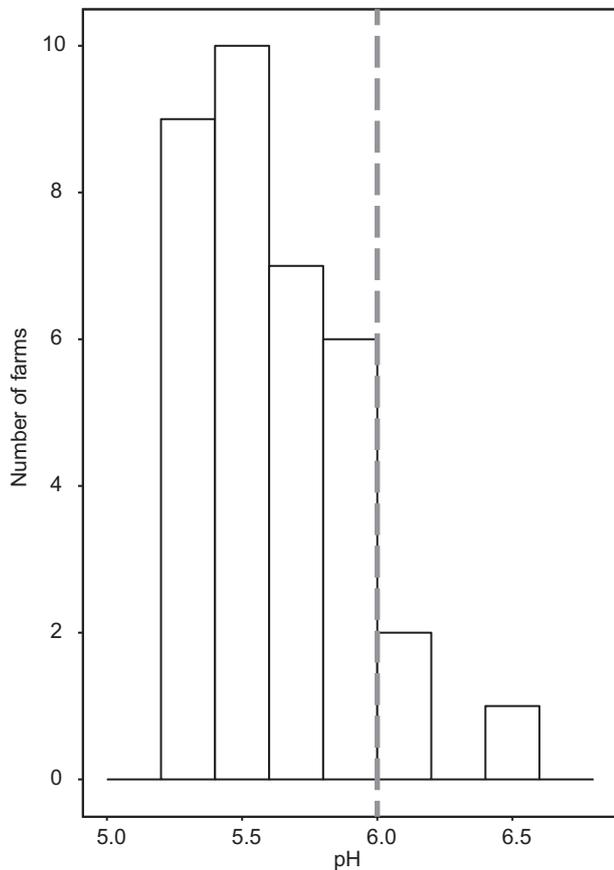


Fig. 3. Mean farm soil pH for sampled fields ($N=35$ farms; two fields per farm). The dashed grey line is the recommended optimal pH for grassland production.

sales representatives from fertilizer companies. Overall, 25 farmers used no external source of advice, 16 farmers took advice from one source and 8 farmers from two sources. Those farmers not using external advice instead used both historical practice and their own knowledge to make management decisions. However, there was no clear relationship between nutrient surplus by advice type (one-way ANOVA for all nutrients, $p > 0.05$).

3.5. Scaling up results to estimate N surpluses and losses at county level

As we have shown that the farms sampled were representative of the population of farms at the county level, the mean annual N surplus from grassland agriculture can be estimated as 5410 t (95% confidence interval estimate from Monte Carlo sensitivity analysis 2636–8265) and the total lime requirement for optimum soil pH as 175,837 t. Assuming $33 \pm 20\%$ of surplus N is leached as NO_3^- (Watson and Atkinson, 1999) and mean Conwy catchment flow, mean NO_3^- -N concentration in leachate from grassland agriculture in Conwy is estimated to be 14.3 mg l^{-1} (95% CI 6.7–23.1). The equivalent fertilizer cost of the annual N surplus at the county level was \$449,103 and the cost of the lime requirement was \$3,955,222 (based on Nix (2009) costs).

3.6. Trade-off between reduced N pollution and greenhouse gas emissions with liming

The estimated additional grass N uptake due to liming for optimum soil pH is estimated as 8.1 kg ha^{-1} (SE 4.19; Stevens and Laughlin, 1996). This would translate into a reduction of 1.1 mg l^{-1} of NO_3^- -N (95% CI 0.0–2.3) in leachate and $0.49 \text{ kg ha}^{-1} \text{ y}^{-1}$ of

direct and indirect N_2O (95% CI –0.01 to 0.95) emissions (including emissions from fertilizer manufacture), i.e. an emissions reduction of $144.2 \text{ kg ha}^{-1} \text{ y}^{-1}$ CO_2 -equivalent (95% CI –2.7 to 283.4). Including transport costs, the estimated CO_2 -equivalent footprint from lime addition is 394% (95% CI 201–21,232) of the estimated savings in emissions from reductions in the sum of emissions from N-fertilizer manufacture and soil N_2O efflux. Hence, under the assumptions here, whilst liming to optimal soil pH for production would reduce N leaching into fresh water, it would increase net GHG emissions at the county level.

3.7. Cost-effectiveness of substituting N application with lime

The net private cost (cost of lime addition less the N-input savings) to farmers of liming to optimum pH every 5 y would be $\$5.84 \text{ farm ha}^{-1}$ (95% CI 1.08–10.78; mean \$432,540 at the county level), while the mitigation cost for the additional emissions would be $\$11.41 \text{ farm ha}^{-1}$ (95% CI 7.70–15.37; \$844,459 county level) for traded C, giving a net-social cost of $\$17.25 \text{ farm ha}^{-1}$ (95% CI 8.80–26.15; \$1,276,989 county level). In a sensitivity analysis exploring how the length of time that applied lime is effective in maintaining soil pH at 6.0 affects farm economics and CO_2 emissions, on average there was a private economic benefit to farmers of liming if it was effective for at least approximately 11 y (Fig. 4). However, there was no net saving of greenhouse gas emissions even if the applied lime was effective for 15 y (Fig. 4). Nonetheless, parameter uncertainty means that the farm cost is not significantly different from zero when applied lime is effective for 6 y or more, and the net social costs are not significantly different from zero when applied lime is effective for 8 y or more. Even with this uncertainty, the model shows there to be a feasible region where farmers have a financial incentive to lime with negative consequences of increased net GHG emissions. At current prices, this region is outside a realistic effective period for liming but any increase in the relative cost of fertilizer compared with lime would increase the incentive to apply lime.

4. Discussion

Predicted restriction in future supplies and increased cost of NPK fertilizers mean that there is an increasing need to maximize nutrient-use efficiency and reduce losses to air and water (with their associated adverse environmental impacts). Sustainable agricultural production must provide a balance between nutrient inputs and outputs, but this is not currently feasible because efficiencies of fertilizer use are poor (ca. 50% for N and <30% for P), leading to surpluses and inevitable losses (Dungait et al., 2012; Swaney et al., 2012). The farm systems associated with the largest surpluses are often intensive livestock farms, especially where the nutrient value of recycled manures is not taken into account in deciding on fertilizer recommendations (Domburg et al., 2000). Surpluses are important because they govern the potential losses that may arise, and farms with a high surplus are more likely to generate larger losses on average (Oborn et al., 2003). However, actual losses are also highly dependent on how the surplus is managed in relation to the prevailing climate and landscape factors that influence vulnerability to loss processes and transport rates to the watercourse (Withers and Lord, 2002). Whilst urine patches are considered to represent the main source of N loss from livestock agriculture (de Klein and Monaghan, 2011), N inputs via fertilizer and manure applications are also key sources of N loss (Beegle et al., 2000; de Klein and Monaghan, 2011). The better use of N via improved fertilizer (Brown et al., 2005) and manure (Beegle et al., 2000; Jarvis, 2011) management are far more pragmatic approaches to increase N-use efficiency than the management of

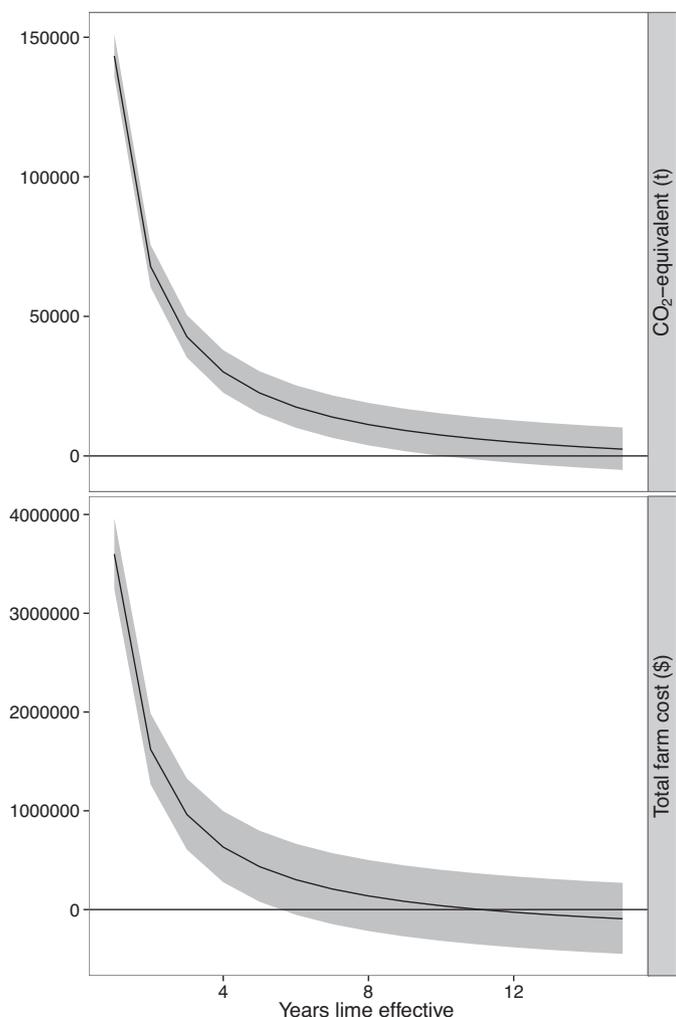


Fig. 4. Sensitivity analysis of county-level results to model parameters of lime application and the time period for which the applied lime is effective at maintaining soil pH at 6.0. The grey region is a 95% confidence interval. Private farm costs become negative (indicating a positive incentive to lime) when its effectiveness is for 11 y or more, while net greenhouse gas emissions (CO₂-equivalents) do not become negative over a 15-y period.

urinary N deposition during grazing and, importantly, are ones that farmers are likely to be more willing (and able) to act upon. Soil, crop, and livestock type, stocking density and nutrient management, as well as the amount of surplus, are therefore vitally important in maximising nutrient-use efficiency and minimizing adverse environmental impacts. In the current study, we sampled only the within-farm range of soil pH and nutrient status (indicated by 'best' and 'worst' fields in terms of livestock production). While the results suggest that the pH of the two fields sampled were similar for the majority of farms, and that the nutrient variation was moderate, it is possible that the fields differed by factors other than pH or nutrient status (e.g. compaction). More extensive sampling would be required to fully capture the within-farm variability.

In the current study, surpluses of N, P and K were small relative to national benchmarks and typical of the more extensive beef and sheep farming on upland pastures in UK with modest fertilizer inputs. The results were largely comparable with indicators for other North-Western European countries. The surplus for P was very low relative to the large proportion of soils with high P fertility, and reflects recent reduction in use of P fertilizers, which have risen sharply in cost. Recent data on fertilizer use show that P and K fertilizer use in the UK has dropped by over 50% in the

last 10 y (AIC, 2011). The high proportion of fields (85%) that were shown by soil analysis to have a P index ≥ 2 is therefore likely to be a cumulative result of past P inputs rather than reflecting the current balance of annual applications and removal in crop uptake. The low current annual surplus of K is more consistent with the high proportion of soils found to have below-optimum levels of K. This difference is attributable to K being taken up in much larger quantities on grassland farms than P, and soil K levels tending to fluctuate more rapidly than soil P levels. It is also noteworthy that surpluses for N and K exceed the rate of fertilizer input, whereas for P they do not. This reflects the high N:P and K:P ratios in livestock manures (Defra, 2010). Optimal crop production is also highly dependent on a balanced supply of nutrients so that one or more elements do not become limiting and affect the efficiency of use of other nutrients (Vitousek et al., 2010). The field testing results suggest that both soil acidity and available K may be limiting crop production in Conwy and reducing the efficiency of utilization of N, with increased risk of N loss. Although our study was conducted in one region, the results are applicable to all agricultural systems, since the need for better soil and nutrient management to optimize productivity has been shown globally (e.g. Goulding et al., 2008; Brown et al., 2005; Koelsch, 2005; Defra, 2010). One way to facilitate this would be to make use of accurate decision-support systems (DSS); however it was of interest that none of the farmers made use of such tools. The majority of farmers were not aware of the existence of these tools, even those such as PLANET or MANNER that are specifically targeted at farmers and frequently mentioned in the farming press (e.g. Clarke, 2013). Matthews et al. (2008) suggest three reasons for the lack of adoption of DSS tools in general, a lack of credibility, resistance to ceding control over decision making and competition from consultants and advisors. In this case the third reason seems the best explanation for the farmers taking advice from other sources and the second reason for the farmers taking no advice. It is not possible to say whether use of existing DSS by farmers would affect the nutrient use efficiency of farms, but if policy-makers are to require farmers to manage the complex trade-off between production, eutrophication and greenhouse gas emissions, more consideration must be given to promoting the tools available. In an alternative perspective, Jakku and Thornburn (2010) highlight the potential value of DSS – not in terms of uptake and use, but as participatory tools enabling learning by farmers and advisors; it seems likely that nutrient management DSS could also serve this purpose.

As mentioned previously, the mechanism for increased N uptake and losses following lime addition is complex; however the relatively small increases in soil pH here through liming would not impact greatly on the rate of nitrification and biological fixation (Kemmitt et al., 2005). Within Conwy County, we estimated that liming improved grassland soils to raise the average pH from the current average of 5.4 to the optimum for plant nutrition of 6.0 would reduce the level of N fertilizer addition required to maintain yields, and that this would enable a reduction in CO₂-equivalent GHG emissions associated with N-fertilizer manufacture and use. However, in terms of EU indicators of the agri-environment, the modest benefit in terms of a reduction in the N surplus and consequent reduced risk of nitrate pollution would be outweighed by the increased CO₂ emissions resulting from the decomposition in the soil of the applied lime. This finding is based on the assumption of maintenance of production at reduced N inputs; it is possible that if N inputs were maintained and production increased, then GHG emissions per kg of protein would actually reduce. There is also a possibility that stocking rates may change in response to any increased grass yield; again the likely direction would be a reduction in GHG emissions per kg protein. We acknowledge that soil and ley type will affect grass yield response to lime application (Stevens and Laughlin, 1996) and further work is needed

to determine the effects of addition of lime or N fertilizer on soil organic matter dynamics and recharge over both the short and long term. Furthermore, within-farm variability and parameter uncertainty mean that it would not be appropriate to use the estimation methodology here to estimate effects at the field scale. However, we consider the approach to be useful at the sub-catchment, catchment and regional scales when considering options for mitigating GHG emissions. This could support important future research into the effects of the spatial scale of N fertilizer or lime application on net greenhouse gas emissions.

There are potentially complex trade-offs amongst ecosystem services and agri-environment indicators in this area of policy. Lime addition might still be of net benefit if the benefits of avoided eutrophication of water bodies are substantial enough to outweigh the costs of the increased greenhouse gas emissions. Another benefit of reduced N-loading in freshwater would be a reduction in the need to purify drinking water, with its associated greenhouse gas emissions due to the energy required. On the other hand, catchment liming can increase the concentration of dissolved organic matter (DOM) released from soil to freshwater (Grieve, 1990) and removal of DOM from drinking water prior to human consumption has a high associated GHG burden. In future research, it would be useful to compare our estimate of the net social cost of N-loading reduction using lime with other methods of abating N pollution in similar farming systems. Unlike greenhouse gas emissions, the social costs of N leaching would be a challenge to incorporate in models because they are highly location-specific and depend on how waters downstream of the pollution source are used (Pretty et al., 2003). Annual variation in rainfall will also lead to temporal variation in concentrations and associated costs (Gibbons et al., 2005). In addition, the marginal social cost of additional units of N varies non-linearly with N-loading (e.g. Nikolaidis et al., 1998). Therefore, while there are additional potentially important social benefits of lime addition (especially through reduced leaching) that could be incorporated in the aggregate social costs (e.g. Pretty et al., 2003) it is beyond the scope of analyses such as those in the present paper to achieve this assessment of total net social benefits. A substantial improvement in the capacity for location-specific modelling of water pollution would be required to make this feasible.

5. Conclusions

Nutrient imbalances have negative impacts on both crop production and the environment. Whilst current emphasis may be on pollution prevention from agriculture, there is also a need to maintain minimum soil fertility levels to support appropriate levels of production. We have seen that farmers may be tempted to omit lime, P and K application because of current high prices and this has the potential for negative impacts on both production and the loss of other nutrients such as N. Whilst reliance on residual P reserves can maintain yields on most soils for some years, residual K reserves can only maintain yields on some clay soils that are K releasing. The lack of use of available nutrient budgeting tools by farmers appears to be primarily due to lack of awareness and competition from other sources of advice. Sustainable food production and the complex trade-off between greenhouse gas emissions and eutrophication requires an increase in the use of such tools, or the development of alternatives that are more amenable for farmer-use.

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