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Modelling the Effect of Climate Change on Environmental Pollution Losses from Dairy Systems in the UK

Agustin del Prado[1,2], Anita Shepherd[2], Lianhai Wu[3], Cairistiona Topp[3], Dominic Moran[3], Bert Tolkamp[3] and David Chadwick[2]

So far, there has been a strong emphasis to study the impacts of climate change on agriculture in terms of changes in food production; however, there is increasing evidence that agricultural ecosystems (e.g. livestock) will also be severely affected in terms of other goods and services. For example, patterns and loads of environmental pollution derived from nutrient losses are expected to change dramatically (e.g. increased run-off: Betts et al., 2007).

There have been few studies that explore with a system-based approach the complex interactions between farm inputs, response of system components and inherent site factors that give rise to changes in productivity, environmental pollution losses and agricultural services in future scenarios.

This article describes the methodology and the results of a study to evaluate the effect of climate change only on losses of nitrogen (N) and carbon (C) from grassland-based livestock systems in 10 UK Regional Development Programme (RDP) areas. In order to do so, a modelling framework integrating different models at the crop and farm level was developed and implemented.

Simulated projections suggest that farming systems will undergo different changes in food production and associated nutrient losses depending on different areas and time-slices. Potential trade-offs on other pillars of farm sustainability (e.g. net farm income, biodiversity and soil quality) were simulated and illustrated as an example.

Keywords: Climate change, farming, GHG, NO_3 leaching, sustainability, systems-based simulation

JEL Classification: Q51, Q53, Q54, Q100

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1. Introduction

Global climate change is expected to severely affect services and goods from agricultural ecosystems. Among these services and goods, changes in crop productivity have received most of the scientific attention in Europe (e.g. Olesen and Bindi, 2002). Modelling results suggest that for a range of medium to high latitude sites and crop types, moderate to medium increases in temperature, along with associated carbon dioxide (CO_2) increase and changes in rainfall patterns may lead to beneficial impacts on crop yields (Easterling *et al.*, 2007). In the UK climate change and adaptation are likely to have the greatest effect on livestock farming systems that rely on grazing given that the grass growing season is likely to increase by between 50 days (Scotland) and 90 days (S England) by 2080s (Hopkins and del Prado, 2007). Manure management may also be affected by climate change and adaptation.

Changes in crop productivity are not just affected by direct changes in climate variables (e.g. temperature, water availability, CO_2 atmospheric concentration) but also by alterations in the biogeochemical cycles, carbon (C), nitrogen (N) and phosphorus (P). The complexity of the interactions and feedbacks is large as these cycles are indeed directly affected by the same variables and in some cases some of these cycles are intimately coupled (e.g. C and N). These interactions determine as well the extent to which the different chemical compounds are lost to the wider environment. Emissions from farming systems must be studied at farm scale and with system approaches (del Prado *et al.*, *in press*), as sensitivity analysis shows that most variability within the lifecycle of agricultural products may actually occur within the farming system, i.e. pre-farm gate (Oenema *et al.*, 2003).

There are few models or modelling systems capable of fully exploring the complex interactions between farm inputs, response of system components and inherent site factors on changes in productivity, environmental pollution losses and agricultural services, either for current or for future scenarios. Examples of European models with some of these characteristics and for current farm scenarios were reviewed by Schils *et al.* (2007).

For climatic change scenarios, to date and to our knowledge, although there have been some studies to evaluate the potential farm adaptations needed to climate change (e.g. Gibbons and Ramsden, 2008; Fitzgerald *et al.*, 2009; Reidsma *et al.*, *in press*) there are very few published studies that have evaluated the effects of climate change on losses at the farm scale for dairy systems (Dueri *et al.*, 2007; Rivington *et al.*, 2007).

In this study a simple modelling framework was developed for a dairy farm in the UK. We used one of the farm models reviewed by Schils *et al.* (2007): the $SIMS_{DAIRY}$ modelling framework (del Prado *et al.*, 2006b; del Prado and Scholefield, 2006; del Prado and Scholefield, 2008; del Prado *et al.*, 2009; del Prado *et al.*, *in press*). The $SIMS_{DAIRY}$ modelling framework was modified accordingly to be sensitive to future climate change variables and to link with the inputs of simulated grass production simulated by LEGGRAZE grassland model (Molle *et al.*, 2006).

The modelling framework was applied to a typical intensive dairy farm in the UK and was driven with a set of medium-high climatic scenarios in different UK regions (UKCIP02 Climate Change Scenarios produced by the Tyndall and Hadley Centres for UKCIP) and for 4 time lines (baseline, 2020s, 2050s and 2080s).

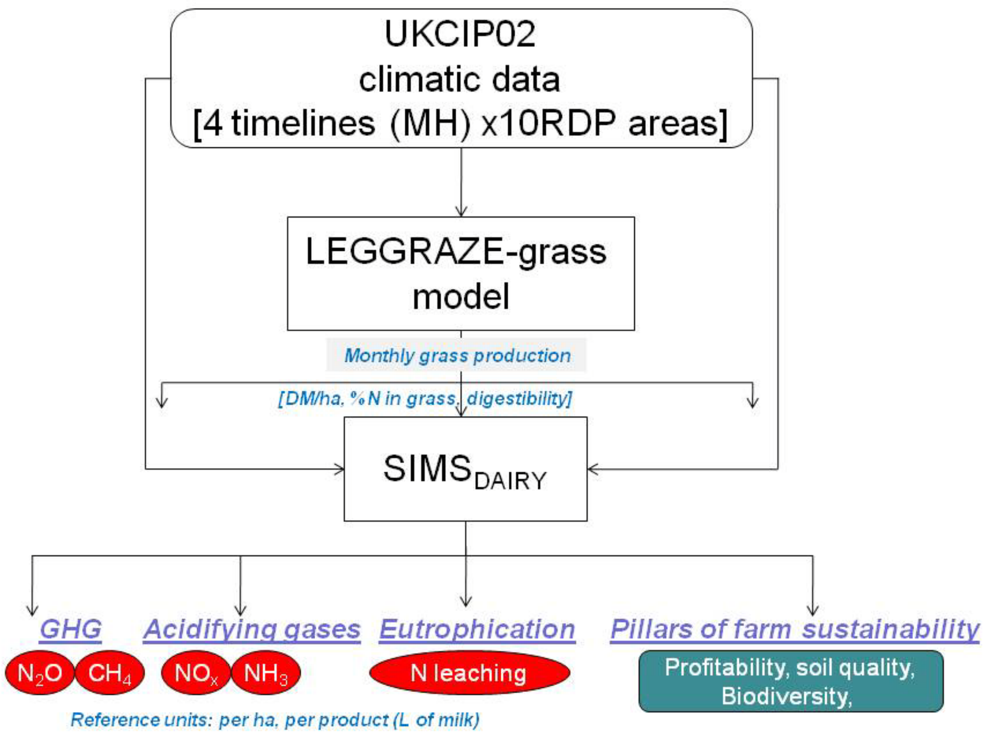
The objective of the study was to explore the effect of climate change only (without adaptations) on a selection of UK pollution losses and selected pillars of dairy farm sustainability. It is expected that climate change will affect different regions and different pollutants in a very different manner.

2. Methodology

2.1. The modelling framework

The framework comprises two main models at the field (LEGGRAZE) and farm scale ($SIMS_{DAIRY}$). Both models were run using the same climatic input data (UKCIP02). The main flows of data between models are shown in Figure 1 (general modelling framework). Monthly data on simulated grass production (dry matter, %N and digestibility in grass) by LEGGRAZE was passed as an input for the $SIMS_{DAIRY}$ modelling framework. The $SIMS_{DAIRY}$ modelling framework simulated for each scenario the effect of climate change on farm losses of nitrous oxide (N_2O), methane (CH_4), ammonia (NH_3), nitric oxide (NO_x), nitrate (NO_3) leaching and on profitability, soil quality and general biodiversity.

Figure 1. General modelling framework with the main input-output data linking LEGGRAZE and $SIMS_{DAIRY}$ models. (Where MH stands for medium-high emission profile scenarios from UKCIP02).



2.2. Brief description of the models

$SIMS_{DAIRY}$: Sustainable and Integrated Management Systems for Dairy Production ($SIMS_{DAIRY}$) is a modelling framework at the farm level which integrates existing models for N and P, equations to simulate NH_3 losses from manure application, predict CH_4 losses and cows' nutrient requirements, 'score matrices' for measuring attributes of biodiversity, landscape aesthetics, product quality, soil quality and animal welfare and an economic model. $SIMS_{DAIRY}$ is capable of simulating farm N and P flows and internal transformations and CO_2 and CH_4 outputs for a given combination of management strategies, soil types and new technologies (e.g. new plant and animal traits: del Prado and Scholefield, 2008). $SIMS_{DAIRY}$ is able to account for pre-farm gate emissions (CO_2-equivalents) associated with purchased concentrates and manufactured inorganic fertilizers (Casey and Holden, 2005) and to reflect the potential change in soil C stocks by adopting a system with higher or lower frequency of cultivation (Dendoncker *et al.*, 2004) and also to account for some CH_4 oxidation by soil (Byrne *et al.*, 2007).

The main assumption of land use in the farm is that on-farm forage is grass and maize. Within grasslands three types of land subtypes are allowed: grass for cutting-only, grass for dairy grazing (and cutting if required) and grass for young cattle grazing. $SIMS_{DAIRY}$´s typical inputs include: average milk yield and concentration (N and butterfat) target per dairy cow, numbers of dairy and young cattle, mineral fertiliser N and P rate in each land subtype, housing period, % of manure applied to each land subtype and timing and site conditions (soil type, history of the field, sward age, climatology). For a complete set of required inputs please see DEFRA project IS0214: final report by del Prado *et al.* (2009).

The interaction between soil type and climate does not only affect processes that directly result in N losses but also on processes regulating the temporal availability of N in the soil and thereby, the processes regulating the pathways of N in the soil-plant-animal system.

Some changes were carried out within $SIMS_{DAIRY}$ to allow inputs from LEGGRAZE model. Herbage production outputs from SAC models for each site and time-slices were extracted and incorporated into input files for $SIMS_{DAIRY}$. Grass variables of monthly DM/ha production, %N content (derived from CP) and digestibility were used for this. The dairy systems assumed that a % of animal diet came from maize forage grown on-farm. As a difference to grass simulation, changes in maize production were simulated by the $SIMS_{DAIRY}$ maize submodel itself.

The new version developed for this study (Figure 2) incorporated the possibility to enter daily climatic data from the UKCIP02 within the code. It also incorporated a new water balance model based on Shepherd *et al.* (2002). It was written to predict the proportion of water stored in the root zone. It is therefore capable of determining % water-filled pore space soil moisture status and also drainage and runoff from soil.

Water availability in the root zone is determined by a water balance in which rainfall, runoff and evapo-transpiration are the main factors. Processes that are affected by both soil temperature and soil moisture content were simulated with new equations. These equations were built to simulate the effect of soil temperature and soil moisture on parameters for processes such as mineralization, denitrification, nitrification, NO_3 leaching, volatilization of NH_3 from different sources of N applied to the soil. The leaching sub-model had to be modified to let drain processes occur during months other than the winter season.

A full socio-economic set of scenarios was not implemented in this study. For example, vast simplifications were taken into account; for example we assumed that the total milk production would not change in time and hence, changes in productivity of milk per unit of land area will result in adjustment of land area used for the farms. The variables for running the economic submodel were also meant to be used as examples.

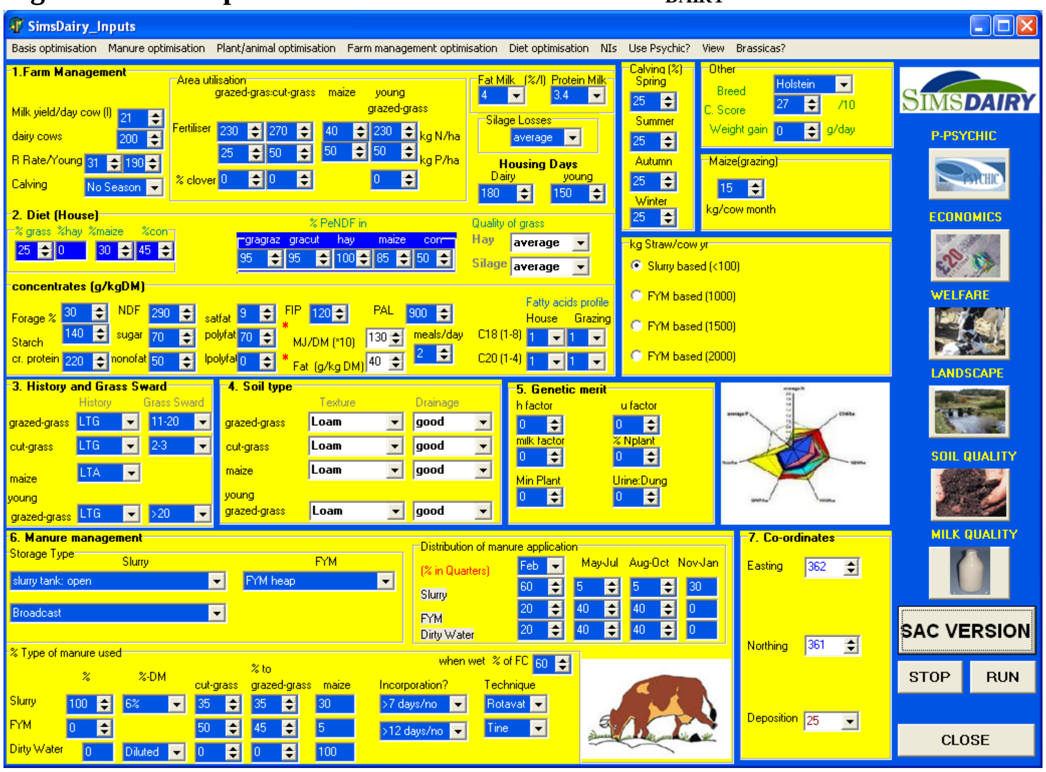
$SIMS_{DAIRY}$ is capable of putting a value to some pillar of farm sustainability. For example, in this study we simulated the effect of climate change on indices of soil quality and biodiversity. The submodel $SIMS_{SCORE}$ assigns a score for each of the sustainability attributes reflecting poor (0) to very satisfactory (6) sustainability.

The soil quality component in the $SIMS_{DAIRY}$ modelling framework refers only to soil structure and some general aspects of chemical fertility. $SIMS_{DAIRY}$ calculates a score index of soil quality based on weighting 4 score sub-indices resulting from assessing: (i) poaching risk, (ii) compaction risk, (iii) erosion risk, (iv) general soil structure and chemical fertility. The general soil structure and chemical fertility sub-index for example is calculated as a function of cultivation, manure and mineral fertiliser management and, the amount of N and P from animal and plant turnover. Soil chemical fertility is positively influenced by large additions of N and P from fertiliser management and animal and plant turnover.

The score index of sustainability for overall biodiversity is based on management factors. A relationship by Herrmann *et al.* (2003) between nutrient input (mineral fertiliser and manure) and biodiversity is incorporated into $SIMS_{DAIRY}$. Six classes (a modification from Herrmann *et al.* 2003 classes) comprising ranges of nutrient inputs (mineral fertiliser + manures) relate the effect of N and P inputs to the soil and overall field-scale biodiversity. Positive scores are also given for cutting for hay. Factors related to soil quality are also linked to the score for biodiversity. Changes in soil quality are likely to have both direct and indirect consequences for grassland fauna and biodiversity in general. For a complete description of how these and other indices are calculated please see DEFRA project IS0214: final report by del Prado *et al.* (2009).

The economic submodel simulates the revenue and costs attributed to dairy farming in the UK. At its core are a series of econometric relationships that replicate the underlying production and cost structures of dairy farm management.

Figure 2. Main input screen for new version of $SIMS_{DAIRY}$.



LEGGRAZE: The model has been used to predict the impact of climate change on grassland production used for livestock production within the UK. The model of the sward developed allows both legume and grass-legume mixtures to be explored. Forage production is calculated on a daily basis, and is presumed to be dependent on herbage mass, temperature, radiation, atmospheric CO2 concentration, available water and nutrients. The grass and legume components within the model are distinguished separately and are divided into leaf, stem, root and dead material.

2.3. Scenarios: climatic, regions, runs and farm typology

Using the IPCC profiles, the UKCIP has created four alternative descriptions of how the climate of the UK might evolve over the course of this century using output generated by the HadRM3 regional climate model (UKCIP02). For each of the four UKCIP02 scenarios, changes are described for three future thirty-year time-slices: 2011 to 2040 (the 2020s), 2041 to 2070 (the 2050s) and 2071 to 2100 (the 2080s). All changes in climate are given relative to the baseline period of 1961 to 1990. Changes to the UK climate are reported across a grid with 50 km cell size. The framework was run for each of the Rural Development Programme (RDP) regions in England plus Scotland and Wales (see table 1).

Table 1. Typical soil types associated to beef, sheep and dairy livestock systems in 10 RDP regions of the UK.

RDP regions	Typical soil type
EE: East England	Clay Loam
EM: East Midlands	Loam
NE: North East	Clay Loam
NW: North West	Clay Loam
SC: Scotland	Clay Loam
SE: South East	Clay Loam
SW: South West	Clay Loam
WA: Wales	Loam
WM: West Midlands	Clay Loam
YH: Yorkshire and Humberside	Clay Loam

In addition to temporal downscaling, June agricultural census data for 2003 was used to identify separate dominant areas within each RDP for dairy (10 x 25 km^2). Using the UKCIP02 medium-high scenario, the weather generator Earwig (Kilsby *et al.*, 2007) was used to generate daily climate date for a period of 50 years for each of the livestock type zones within the RDP and for each of the climate change time slices.

For the selected areas in England and Wales we defined the dominant soil types based on the soilscapes data (htpp://www.landis.org.uk). The resulting soil types (textures) are shown in table 1.

We ran $SIMS_{DAIRY}$ for the combination of 10 RDP areas x 4 time slices x 30 years. Averages from 30 years of losses of N and C were recorded and analysed. Results were set to adjust the hectares needed for forage instead of adjusting the number of cows or level of milk production that certain reductions changes in plant production per hectare could trigger. In total over > 3,000 model runs were completed. Other impacts affecting the overall sustainability of the dairy farm were also recorded from the simulated results of $SIMS_{DAIRY}$ (e.g. net farm income, land area required, biodiversity index and soil quality index).

We defined a baseline typical dairy farm in the UK. This baseline farm had typical fertilizer application rates for each grassland area and different proportion of total area used for grazed grass, cut grass (different number of cuts was also taken into account) and production of maize silage. The main characteristics of the farm are shown in Table 2, which describes key components of a conventional dairy farm which typically relies on on-farm grass and maize production and bought-in concentrates for sustaining animals.

Table 2. Main characteristics of the typical dairy farm used as baseline.

Farm management				
Milk yield (litres/ cow yr)	7600			
Fat in milk (g/kg)	40			
Protein in milk (g/kg)	34			
Dairy cows (number)	200			
Replacement rate (%)	31			
Followers (number)	190			
Calving pattern	All-year			
Breed	Holstein			
Silage management	Average quality			
Housing time-Dairy cows (days/year)	180			
Housing time-Followers (days/year)	150			
Diet				
During housing	grass silage, maize silage, concentrates			
During grazing	grazed grass, maize silage, concentrates			
Annual Fertiliser management				
		grass		***maize***
	cut	grazed (dairy)	grazed (followers)	
Fertiliser N (kg N/ha)	230	270	230	40
Manure management				
Type of manure	slurry (60 g /kg DM*)			
	cut-grass	***grazed-grass***	***maize***	
% of total applied to land	35%	35%	30%	
Storage	slurry tank: open			
Application technique	Broadcast			
Grassland management				
	cut-grass	***grazed-grass***	***young grazed-grass***	
History		Long term grassland		
Sward age (years)	2-3	11-20	>20	

*DM: dry matter

Some assumptions were made to the dairy system in order to simplify the simulations. For instance, any cattle other than lactating dairy cows was simulated with the assumption that it would be represented by an average follower of a bodyweight size of 300 kg with no body-weight change during the year. The grassland area was split into cut-only fields (3 cuts for silage) and grazed fields (with 1 small cut generally). The timing and percentage of annual mineral fertiliser applied per month was designed to follow the UK fertiliser recommendations for agricultural crops (RB209), (MAFF, 2000) and timing for the manure applied to land followed the distribution patterns described by Smith *et al.* (2001), in which, the proportion of manure applied of the annual total is as follows: from February-April (40%), May-July (10%), August-October (25%) and November-January (25%).

Seasonal milk production can have substantial effects on not only the economics of the farm but also the seasonal requirements for the herd and thereby, it may affect the needs of feed supply from varied sources (e.g. grazed grass *vs* silage). For this study we used an all-year round pattern whereby the dairy herd has equal amount of cows calving every month and thus, on a daily average basis, the same amount of averaged milk is expected at any time of the year.

3. Results

The results corresponding to gaseous pollutants per L of milk [overall GHG (as CO_2 equivalents global warming potential), N_2O, CH_4, CO_2, NH_3, NO_x], soil C storage and some indices reflecting farm sustainability (net farm income, milk production per hectare, land area, level of biodiversity and soil quality) are shown in Figure 3 as radar graphs. It must be noted that Figure 3 results are based on averaging 180 data points resulting from running $SIMS_{DAIRY}$ for a combination of 30 years per time slice x 6 summer grazing regimes (varying the % of total cattle DM intake coming from grazed grass during the grazing period). The predicted results from the baseline scenarios were transformed to be 1 and the predicted results for the different time lines were transformed proportionally to this value of 1. Only values < 1 implied improving the results from the baseline scenarios. Average NO_3 in the leachate and NO_3 leaching per hectare simulated results were included in a different table (Table 3) since these values varied substantially in time and they could have produced a distorted image of the spider plots. For a full description of the absolute values for all of the variables studied see the final project report by Moran *et al.* (2009).

3.1. Greenhouse gas (GHG) emissions

Greenhouse gas emissions greatly varied depending on the specific gas, site and time-slice. For the baseline year, overall N_2O emissions varied between 6.6 kg N_2O-N/ha yr and 22.6 N_2O-N/ha yr for the SE and SW regions, respectively (*data not shown*).

Most future time-slices scenarios resulted in decreasing N_2O emissions, expressed per ha (data not shown) and per unit of milk (Figure 3). The greatest reductions would occur in the WM region (40, 50 and 80% as N_2O emissions per L of milk for 2020s, 2050s and 2080s, respectively). NW, SW, EE, SE and EM regions would also reduce substantially their emissions of N_2O (up to 72, 68, 66, 62 and 60% reduction, respectively). These decreases were generally incremental in time. The most modest decrease took place in the YH region where a maximum reduction of N_2O per L of milk was achieved in 2050s (25%).

For the baseline year, overall CH_4 emissions varied between 193 to 282 kg CH_4/ha yr for the WA and SW regions, respectively (data not shown). Overall CH_4 emissions per L of milk were similar at all sites and time-slices ca. 27 g CH_4/L of milk. Pre-farm gate CO_2 emissions generally decreased in time. The maximum reduction in time was at WA where CO_2 emissions per L of milk were reduced approximately 10% (Figure 3).

All future scenarios resulted in both an increase in CH_4 emissions/ha (data not shown) and no change in CH_4/L of milk (Figure 3). These values were completely related to the amount of forage area required for each scenario.

For the baseline year, overall GHG emissions (CH_4+N_2O) expressed as global warming potential as C equivalents varied between 7819 to 14056 kg C-eq/ha for the SC and SW regions, respectively (data not shown).Most future scenarios resulted in a decrease in C-eq emissions expressed per litre of milk (Figure 3) and per hectare. Carbon-eq emissions per litre of milk were always reduced with time. For example, C-eq emissions from the SW, WM and NW regions were reduced by up to 34, 32 and 31%, respectively (Figure 3). Reductions of C-eq emissions per litre of milk were smallest at WA and YH (up to 6% and 15% for 2080s, respectively).

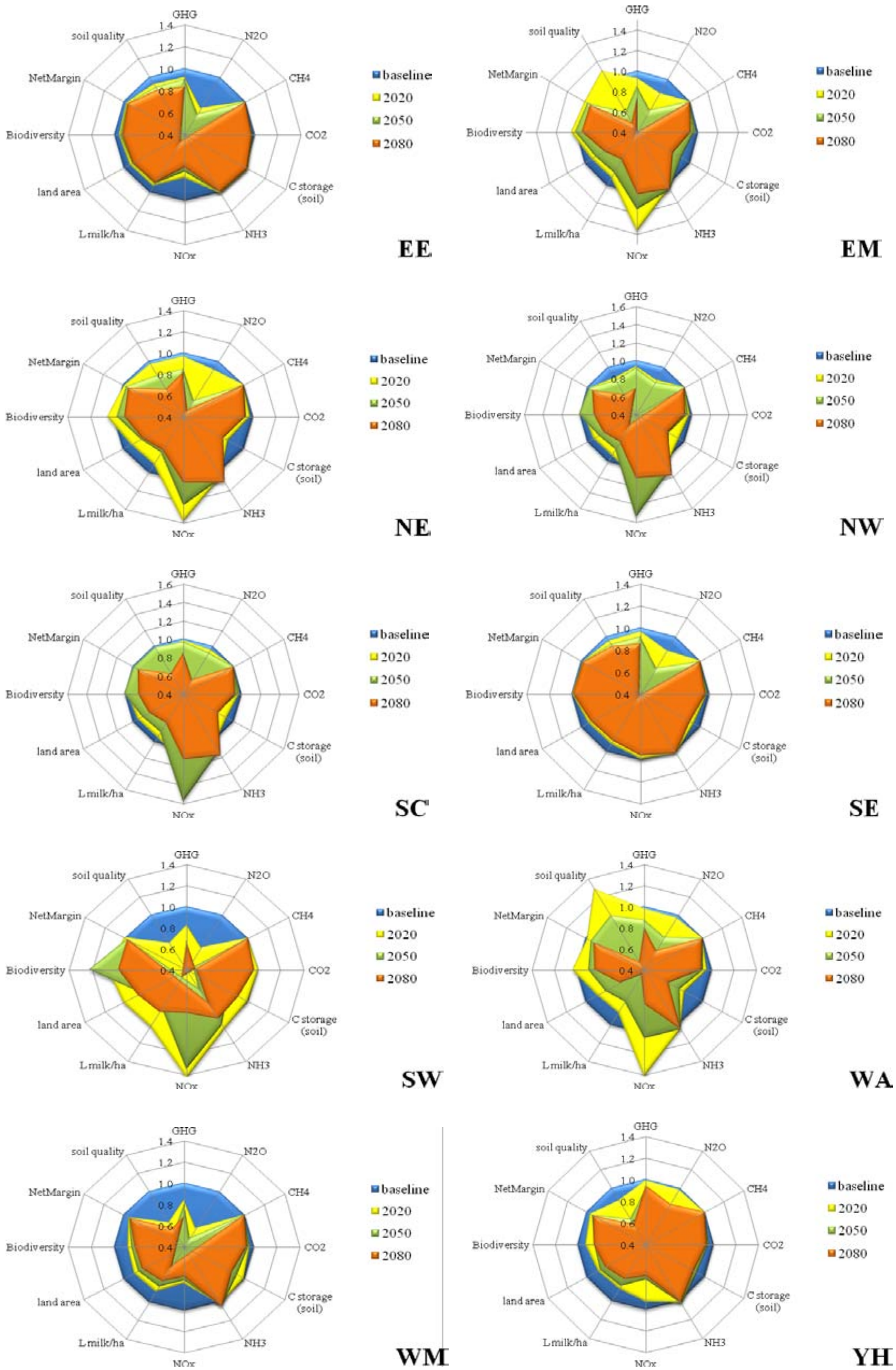
3.2. Acidifying gases

Ammonia and NO_x emissions greatly varied depending on the specific gas, site and time-slice. For the baseline year, overall NH_3 emissions varied between 37.8 kg NH_3-N/ha yr (WA) and 64.1 NH_3-N/ha yr (EE) (data not shown).

Simulated climate change scenarios resulted in varied trends in NH_3 emissions. In most regions, NH_3 emissions increased both expressed per ha and per L of milk. The largest increase of NH_3 emissions per L of milk, for example, was found at NW and SC with increases up to 18 and 17%, respectively (Figure 3). A decrease in NH_3 emissions per L of milk was only found in the SW and this only happened in the 2080s (up to 6% reduction) (Figure 3). Very small changes in time were predicted in the SE and EE as NH_3 emissions per L of milk (Figure 3).

For the baseline year, overall NO_x emissions varied between 1.2 to 2.7 kg NO_x-N/ha yr for the WA and SW, respectively (data not shown). Future scenarios resulted in varied trends in NO_x emissions, no clear trend could be established for the whole set of RDP areas in time. Reductions in NO_x/L of milk were for example found in the WM, YH, EE and SE regions (Figure 3). Simulated values in the 2080s were reduced by up to 31%, 31%, 30% and 4% for the aforementioned areas, respectively. In the WA and SW regions, emissions of NO_x/L of milk were reduced only in the 2080s (Figure 3). The rest of the regions showed increasing NO_x emissions. Largest increases in the 2080s were in the NW and SC regions (up to 11% for both areas) (Figure 3).

Figure 3. Comparison between baseline and 2020s, 2050s, 2080s for variables indicating environmental pollutants and variables indicating other attributes of sustainability of a dairy farm .



3.3. Eutrophication losses

For the baseline year, average concentration in the leachate varied between 3.3 (SW) and 49 (EE) mg NO_3-N/l (data not shown). Results are based on averaging 180 data points resulting from running $SIMS_{DAIRY}$ for a combination of 30 years per time slice x 6 summer grazing regimes.

Nitrate leaching per hectare for the baseline year ranged from 35.3 kg N/ha yr to 76.8 kg N/ha yr for SW and EE, respectively (data not shown). Nitrate leaching losses varied substantially between sites and time-slices. Predicted change (%) in NO_3 leached (as concentration in the leachate and per unit of hectare) is shown in table 3.

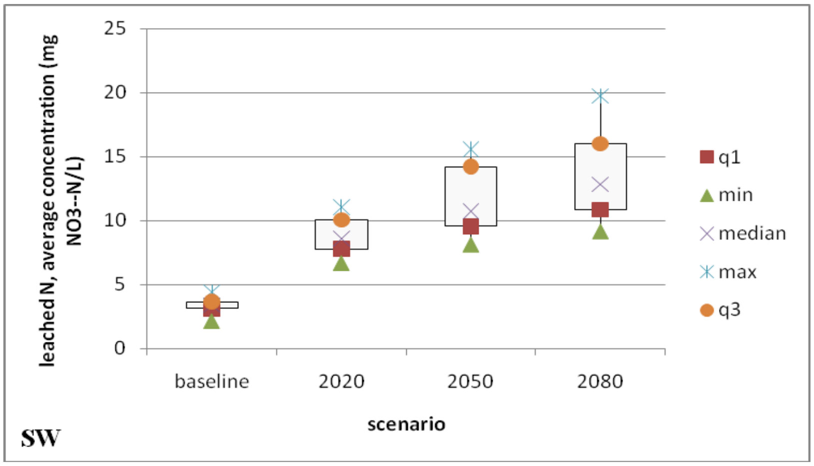
Table 3. Predicted change (%) in NO_3 leached (as concentration in the leachate and per unit of hectare) for the different RDP regions and with projections of climate change in 3 future time slices.

Region	EE	EM	NE	NW	SC	SE	SW	WA	WM	YH
					Mean concentration in the leachate of NO_3 (mg/l)					
2020	27%	79%	82%	2%	3%	14%	163%	28%	51%	9%
2050	16%	109%	117%	120%	52%	20%	245%	7%	110%	49%
2080	32%	157%	99%	87%	74%	41%	303%	10%	135%	51%
					NO_3/ha					
2020	0%	48%	19%	-10%	-7%	6%	135%	11%	1%	-2%
2050	1%	44%	28%	27%	15%	-10%	155%	-19%	8%	1%
2080	6%	32%	20%	11%	1%	15%	153%	-50%	18%	0%

Average N concentration in the leachate consistently increased with time. Large increases occurred at SW (up to 303% at 2080s), EM (up to 157% at 2080s) and WM (up to 135% at 2080s). The smallest increases were found at WA site (up to 10% at 2080s). These increases were consistent with the decrease in annual drainage volume in the future scenarios. Nitrate leaching per hectare increased with time in almost all sites except for WA site, where a decrease of up to 50% was found in 2080s. The largest increase in N_{O3} leaching was at SW site (>150% in both 2050s and 2080s). The smallest changes in NO_3 leaching were found at YH and EE areas with 0% and 6% increase in 2080s, respectively.

Some RDP regions showed a large range of NO_3 leaching results. Given the deterministic nature of $SIMS_{DAIRY}$ these ranges are caused by large climatic differences within years of each time slice simulated. Two graphical examples are provided as a simple way to show these ranges. Figure 4 illustrate, in the form of a "box and whiskers" plot, the variation in mean annual NO_3 concentration in the leachate for the 4 time slices in the SW area. Figure 5 illustrates the frequency distribution of annual NO_3 concentration in the leachate for the 4 time slices in the NE area.

Figure 4. Median and range (defined by minimum, maximum and upper/lower quartiles of 180 data points) simulated values of mean annual NO_3 concentration in the leachate for the 4 time slices in the SW area.



Variability in mean NO_3 concentration in the leachate seems to increase with time in most cases, for example in SW (Figure 4) and NE (Figure 5). It must be noted that for the NE area and for 2020s time slices (Figure 3) a very large range in mean NO_3 concentration in the leachate was simulated.

Figure 5. Frequency distribution (%) of annual mean NO_3 concentration in the leachate for the baseline and 2020 time slices in the NE area. Results are based on 180 data points.

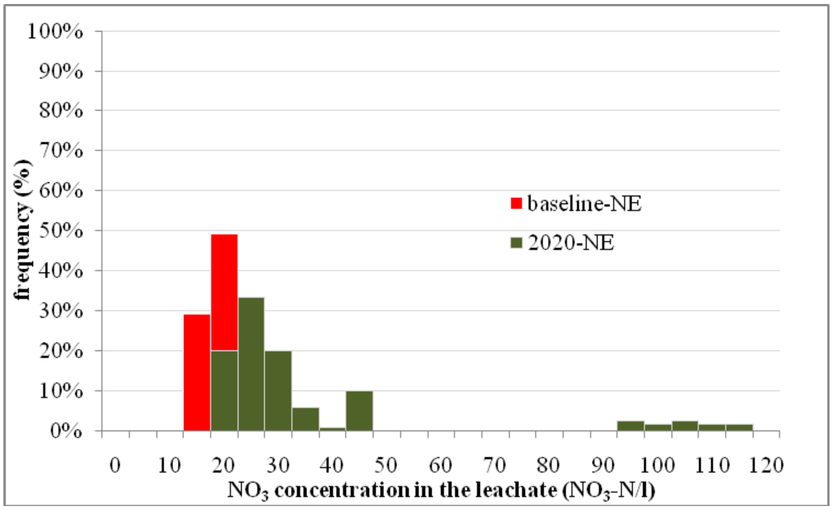
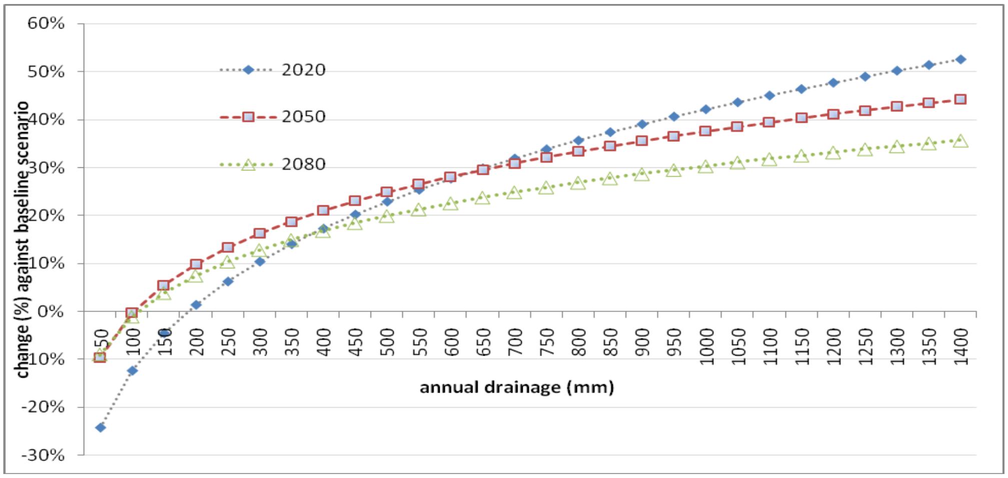


Figure 6 shows the clear relationship between annual drainage volume and % changes in average NO_3-N concentration in the leachate for the 3 future time slices. The absolute values in relation to these changes can in fact be fitted to a power equation with a high and positive correlation ($r > 0.9$ and S.E = 1.1, 95 % confidence: data not shown). The larger the annual drainage volume the smaller the average NO_3-N concentration in the leachate.

Figure 6. Relationship between annual drainage volume and changes in average NO_3-N concentration in the leachate in relation to the baseline scenario.



3.4. Potential trade-offs on other pillars of farm sustainability

The projections of different indicators of potential trade-offs or impacts on other aspects of dairy farm sustainability are shown in Figure 3. The following indicators are shown: milk yield/ha, land forage area (grass + maize) required in dairy systems, net farm income, biodiversity and soil quality.

For the baseline year, net farm income values ranged between 48506 (SC) and 52709 £ (SW) (data not shown). The largest incomes were found in SE and SW sites, which coincided with the sites with greatest milk production per hectare (data not shown). Future scenarios brought about an increase in total net farm income possibly caused by the increase productivity of the land and consequently a smaller requirement of forage land area to produce the same amount of milk than in the baseline scenario.

For the baseline year, the biodiversity index varied between 0 (WA) to 1 (SE), which reflects the low level of potential biodiversity for intensive dairy farming systems. Future scenarios showed differences in trends between sites. Most areas resulted in an increase in biodiversity level in 2080s, however, some decreased the level up to 2050s (e.g. SW: up to 22% decrease) and some other increase the level of biodiversity for all time-slices (e.g. WM and YH).

For the baseline year, the soil index varied between 1 (SW) to 2 (EE). Soil quality values were in relation to soil compaction, erosion and poaching. Most differences were related to the climatic changes and the rainfall patterns. Soil quality index decreased for the 2020s time slices in EM and WA RDP areas. All areas resulted in an overall increase in soil quality level in 2080s. The largest increase was at SW area (133%) and the smallest was at SE area (10%).

3. Discussion

As expected, changing climatic conditions greatly affected N losses. A combination of 2 factors was responsible for controlling such losses. First, different processes compete for the available N in the soil, and second, soil water conditions greatly regulate the oxidative level of the N lost (e.g. as NO, N_2O, N_2). Nitrate may: (1) undergo denitrification to gaseous oxides of N and to N_2, (2) be taken up by organisms (assimilatory reduction), (3) be used by microorganisms as an electron acceptor and become reduced to NH_4^+ (dissimilatory reduction), (4) be lost in leachate or run off, or (5) accumulate in the soil. Ammonium may: (1) be taken up by plants, (2) be immobilised in microbial biomass, (3) nitrify to NO_3^-

and partially be lost as gaseous oxides of N (4) be leached, (5) accumulate in the soil (Paul and Clark, 1996) or (6) volatilised as ammonia (NH_3). The most important factor affecting those pathways is the increase in plant production per unit of both land area (ha) and kg of animal product (data not shown). Even though that mineralization rates are expected to increase with warmer and wetter climates and thereby a greater amount of inorganic N is expected to be released in the soil from the soil organic matter, the increasing plant N uptake in future climate change scenarios still caused a reduction in soil inorganic N available for the rest of the competing processes leading to loss of N to the wider environment.

The form of N that is lost via denitrification and nitrification is greatly regulated by soil water content. The rate of NO emission is greater with drier soils and decreases as the soil moisture content approaches field capacity. As the soil atmosphere becomes more O_2-limited, so, N_2O and N_2 emissions increase while emissions of NO decrease. Evidence suggests that the optimum soil % WFPS values for maximum NO emissions are between 30% and 40% WFPS (e.g. del Prado *et al.*, 2006a). Nitrous oxide emissions increase to a level where simultaneous denitrification and nitrification are at their maximum (75% WFPS). Above this soil water content, denitrification is the main process producing N_2O and, as the soil became more anaerobic, emissions of N_2 became greater than those of N_2O.

Nitrate leaching losses were affected by factors that regulate the competition for the available N in the soil during the period previous to autumn-winter drainage season. Among these factors, increasing plant N uptake would result in a decrease in leaching but decreasing denitrification losses through disfavouring soil anaerobic conditions would lead to an increase in soil leachable N for the autumn-winter drainage season. Smaller drainage volumes in future scenario would result in a reduced amount of total N being leached and an increase in the concentration of that N actually leached. In this study, as land area required was adjusted to the amount of food produced, assuming a constant product (e.g. total L of milk in a farm) resulted in greater NO_3 leaching per hectare but fewer per unit of product (or total) (data not shown). A more sophisticated case study where spatial farm units had been explicitly accounted for and integrated within catchments and where socio-economic story lines had been included would have produced a more realistic picture.

Most CH_4 emissions were caused by animal enteric fermentation. Although CH_4 emissions from dung deposition may have varied from region to region and time-slices, they are minor in comparison to enteric fermentation and would not influence the overall CH_4 emission at the farm level. Methane emissions per unit of milk did not vary substantially. Methane output per unit of product was inversely related to %N content as more protein in the diet generally increased the amount of kg of milk per unit of DM ingested by the animal. This study did not account for the potential changes in animal energy use in future time-slice scenarios. The quality of diet was not included as a factor affecting CH_4 emissions from enteric fermentation. Although there is some evidence that some factors (e.g. fat content) may have a strong effect on CH_4 output from enteric fermentation, this study did not cover these alterations of the systems and therefore, no changes were expected to occur in the whole total CH_4 balance. Acidifying gases, especially NH_3 emissions, could potentially be of greater concern in future UK livestock scenarios. Warmer conditions will increase NH_3 loss unless methods of manure injection are used. In fact, this study did not take into consideration the role of existing and potential future environmental regulations to mitigate some of these emissions. For example, NO_3 leaching through the NO_3 Directives or Water Framework Directive is likely to limit further the intensity of nutrient use in livestock farms. Unless more appropriate NO_3 leaching mitigation measures are taken in the future, it is likely that some UK farming areas will be having mean concentrations in the leachate over the NO3 Directive Threshold (50 mg NO_3/l).

The challenge will be to understand how different regulations targeting different pollutants can be integrated in the future. The role of economics at the farm and greater levels will obviously lead the changes in adaptations to climate change and the interactions with the future environmental regulations.
The economic results were simulated by assuming that all input values would not change with geopolitical scenario and site. Due to this simplification, we did not intend to produce a robust assessment of the impact of climate change on the economic returns; however we have used these results as an example of possible differences caused only by climatic changes in different areas of the UK. As Glendining *et al.* (2009) indicated for sustainability-related studies, land area should be also included in any assessment. Land area was included in our study, but an improved methodology would require costing changes in land requirement to produce the same amount of food. Economically-based policies may in turn be in serious conflicts with those more related to environmental risks due to this mismatch of scale relevance. The effectiveness of mitigation methods to decrease any particular pollutant will depend for example on the farmer or land user´s response to any potential economic benefits or penalties due to implementation and in relation to market dynamics.

Co-benefits of increasing the efficiency of N plant use included greater opportunities for improving the botanical diversity. This fact, however, needs further testing as the temporal variation of inorganic N flows in the soil within a year may have different implications for different potential plant species.

Positive changes in soil quality were observed for future climate change scenarios possibly due to a decrease in rain and thereby a decrease in soil poaching and erosion by grazing animals. This fact however needs further testing as although monthly rainfall rates could be smaller in future scenarios and for certain areas; the frequency of intensive storm events will increase. Although the $SIMS_{DAIRY}$ cannot study the soil quality form the biological viewpoint, it can be speculated that climate change is likely to affect biological functions within the soil. New experimental evidence suggest that soil drying ad re-wetting events may actually have a very strong effect on losses such as the release of soil soluble P in the leachate (Blackwell *et al.*, 2009) or N and C gaseous emissions (Muhr *et al.*, 2008). Future versions of SIM_{DAIRY} should include shorter time-steps to pick up these events.

4. Conclusion

As expected climate change (only) affected N (i.e. nitrogen) and C (i.e. carbon) losses derived from N and C cycling in a dairy farm in very different ways depending on the form of N and C studied and site. This is a typical example of the challenge to achieve win-win situations for all of the pillars that define farm sustainability. In general, there will be an increase of the negative impact that intensive dairy farming has on acidifying gases and NO_3 concentration in the waters and a positive effect on the GHG emissions. Farm productivity will increase driven by plant production increase per hectare and this will have a positive effect on the profitability of the farm. Variability in rainfall rates and temperatures still remains the main challenge to predict changes in future climate change scenarios. Future studies should include robust socio-economic assumptions and include mechanisms of integrating mitigation targeting different pollutants with specific potential environmental regulations.

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