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Abstract: This report describes mechanisms of trade-offs and interactions between adaptation and mitigation and between various sources and types of GHG emissions at animal, manure, field and soil level. The impact of climate change is illustrated by show-case farms and maps of European grasslands. Different process-oriented simulation models at animal and field level were used: the Dutch Tier 3 for enteric fermentation, PaSim and DNDC for soil and vegetation, and a newly developed Danish model for manure storage.

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1. Introduction

The EU-project AnimalChange will provide scientific guidance on the integration of adaptation and mitigation objectives and on sustainable development pathways for livestock production in Europe, in Northern and Sub-Saharan Africa and Latin America. Work Package (WP) 8 of AnimalChange (“integrating adaptation and mitigation options”) is targeted at the field and animal scale. In WP8 the implications of adaptation on the potential to mitigate greenhouse gases (GHG) emissions are tested, and vice versa, the implications of mitigation on the potential to adapt to climate change. Mitigation options are options which reduce the net emissions of the GHG carbon dioxide (CO₂), and emissions of methane (CH₄) and nitrous oxide (N₂O) from livestock production systems. Adaptation options describe ways for livestock production systems to adapt to future climatic conditions (such as higher temperatures, larger climatic variability and increased frequency and severity of droughts and floods, but also higher atmospheric concentrations of CO₂ which may have a C fertilisation effect and positive effect on plant growth).

The present deliverable D8.4 describes the mechanisms for trade-offs between sources and sinks of greenhouse gas (GHG) emissions, and identifies the interaction between adaptation and mitigation (greenhouse gas) emissions. Results relate to Tasks 8.2 and 8.3 in WP8 where process-oriented models were used, and if necessary adapted or improved, to evaluate the effect of mitigation measures under various conditions. Task 8.2 was targeted at intensive ruminant production systems and Task 8.3 at extensive, pasture based, ruminant production systems. A previous deliverable D8.2 provided the first version of estimates of mitigation and adaptation options based on process-oriented models (Bannink *et al.*, 2013a). A subsequent deliverable D8.3 included further work on mitigation and adaptation options and evaluations of specific farm cases. This deliverable D8.4 provides results for a selected set of mitigation measures thought feasible for livestock systems under various conditions and under adaptation to climate change (CC) and reports on the mechanisms underlying the effectiveness of mitigation options as influenced by adaptation of livestock systems to CC effects.

For testing the effect of different options and CC scenarios, mechanistic, dynamic models have been used: the Dutch Tier 3 for enteric CH₄ in dairy cows at the animal level (Bannink *et al.*, 2011; 2010), the PaSim (Ben Touhami *et al.*, 2013; Ma *et al.*, 2015; Vuichard *et al.*, 2007a & 2007b) and DNDC models (Li *et al.*, 2010 & 2011) for the field level (in case of PaSim, both field and animal level), and a newly developed model at the manure storage level (Hutchings *et al.*, unpublished). Modelling results are described and discussed for promising options that have been identified in previous workshops, from WP6 (mitigation) and WP7 (adaptation) and from deliverable D8.1, “Qualitative overview of mitigation and adaptation options and their possible synergies and trade-offs” (Van den Pol-van Dasselaar, 2012).

Chapter 2 provides materials and methods for the use of the process-oriented models. Chapter 3 discusses the mechanisms of trade-offs between different sources of GHG in ruminant production systems, dairy farms in particular. These trade-offs are discussed for the various mitigation measures at the field level and animal level, respectively, and adaptation to climate change. Where feasible, reference is made in Chapter 4 to results of a comparison between the GHG emissions for three show-case dairy farms that have been reported in deliverable D8.3. These show-case dairy farms differed widely in the level of intensity of feeding, stocking density and milk yield per hectare, intensity of N fertilization, and extent of purchase of external inputs (ranging from no grazing (stall-feeding) and highly intensive feeding to unrestricted grazing with minimal amounts of dietary supplementation). The three farms were

compared by making the integral use of these process-based models and consequences for calculated on-farm net GHG budget were determined. Chapter 5 discusses the effect of increased proportions of legumes and the introduction of legumes on GHG emissions from European grasslands / grass-based livestock systems under various CC scenarios. Finally, Chapter 6 discusses and integrates the results from preceding chapters, followed by general conclusions in Chapter 7.

2. Materials and methods

2.1. Mechanistic, dynamic models

In WP8 of AnimalChange process-oriented models are used. Adopting generic constants for emission factors (according to IPCC Tier 2 methodology; IPCC, 2006) keeps inventory methodology less complex and more transparent; however, it also ignores variation and does not acknowledge mechanisms underlying this variation. Process-oriented models give insight in this variation. Generic constants can be particularly useful for the purpose they have been derived for, which is hence in principal their use as a generic value and not a case specific value affected by many detailed aspects of farm management. The Tier 2 approach is mostly used for national inventories of GHG emissions. However, for key sources of national GHG emissions IPPC recommends development of Tier 3 approaches. The argument to develop Tier 3 approaches becomes even stronger when the aim is to study variation in these key sources of GHG emissions between different farming conditions. For this reason only such Tier 3, or candidate or Tier 3-like, approaches have been used in the present study to explore the effect of mitigation options and the impact of (adaptation to) climate change on this.

The Dutch Tier 3 model (Bannink *et al.*, 2011) has been used to test the effect of mitigation measures on enteric CH₄ emission. The model requires animal characteristics (feed intake, milk composition) and feed characteristics (dietary chemical composition and intrinsic rumen degradation characteristics) as an input (Dijkstra *et al.*, 1992 & 2008; Mills *et al.*, 2001; Bannink *et al.*, 2006, 2008 & 2011). These inputs largely correspond to those adopted in protein evaluation systems used in current practice. The model was adapted to deliver estimates of manure composition and milk production next to that of enteric CH₄. In this manner the model identifies key aspects of enteric fermentation and enzymatic digestion that need to be taken into account when the aim is to obtain accurate estimates of emission parameters and cow performance under specific feeding and farming conditions.

Variation in emissions from soils, from applied manure (both ruminants and monogastrics) and from excreta of grazing animals are represented in the models PaSim (Vuichard *et al.*, 2007a, b), and DNDC (Li *et al.*, 2011). These models are the state-of-the-art and take into account the large impact of management and environmental conditions on field emissions. Comparable to the model of enteric fermentation and excretion, the model of soil denitrification requires inputs on fractions of organic matter in manure which differ in availability for soil microbiota. The model also requires several meteorological and soil management data as input because these have a major impact on the soil environment where microbial activity and denitrification takes place. In addition, within AnimalChange a model was developed to describe emissions from stored manure (Hutchings *et al.*, unpublished) which also requires inputs for several nitrogenous and carbonaceous fractions.

2.2. Dutch Tier 3 for enteric methane in dairy cows

2.2.1. Model representation, model aim and model use

The basal part of the current Dutch Tier 3 model for enteric CH₄ emission in dairy cattle is a representation of the dynamical aspects of the interaction between feed substrates and microorganisms in the rumen (Dijkstra *et al.*, 1992). Most important factors known to affect microbial activity and feed substrate degradation were included. The model aims to obtain an improved understanding of how feed and animal characteristics and rumen fermentation conditions affect

feed degradation and microbial activity, and the end-products of microbial activity that are absorbed (ammonium, volatile fatty acids) from rumen or flow out to the small intestine (microbial matter and undegraded substrates).

Later versions of the model were made more specific for enteric fermentation in lactating dairy cows by including a representation of the stoichiometry of volatile fatty acid production (Bannink *et al.*, 2006; 2008) and rumen hydrogen balance (Mills *et al.*, 2001; Figure 2.1) that was derived from in vivo data of rumen digestion in lactating cows only. Based on this enteric hydrogen balance, after addition of a representation of fermentation processes in the large intestine comparable to that of the rumen, and under assumption of total conversion of net hydrogen surplus into CH₄, the model calculates enteric CH₄ emission. Empirical equations were added to represent the digestive processes in the small intestine and the outflow of substrate into the large intestine.

The current model version is used to investigate how feed and animal characteristics affect enteric fermentation and digestive processes, and what consequences are to be expected for the amount and profile of nutrients absorbed from the gastrointestinal tract, for excretion and composition of urine and faeces (to be related to total ammoniacal nitrogen and ammonia emission), for the production of milk (given its composition), and for CH₄ emission.

2.2.2. Model structure

The model is a process-oriented model and hence consists of a set of ordinary differential equations that describe the change in time of pools of substrate, micro-organisms and microbial end-product present in the rumen and large intestine (Figure 2.1). The inflows and outflows from these pools are described and parameterized as much as possible from reports of in vivo trials. The model identifies several types and forms of substrates. It makes a distinction between soluble or degraded substrate, potentially degradable substrate, and undegradable substrate. It distinguishes between sugars and starch as amylolytic carbohydrates used by amylolytic micro-organisms, and cell wall material as a carbohydrate source for fibrolytic micro-organisms. The model distinguishes three types of micro-organisms; amylolytic bacteria and fibrolytic bacteria utilizing the carbohydrate sources with retention times of fluid and particulate substrate, respectively, and protozoa that predate on bacteria and have a much longer retention time in the rumen.

2.2.3. Model inputs and outputs

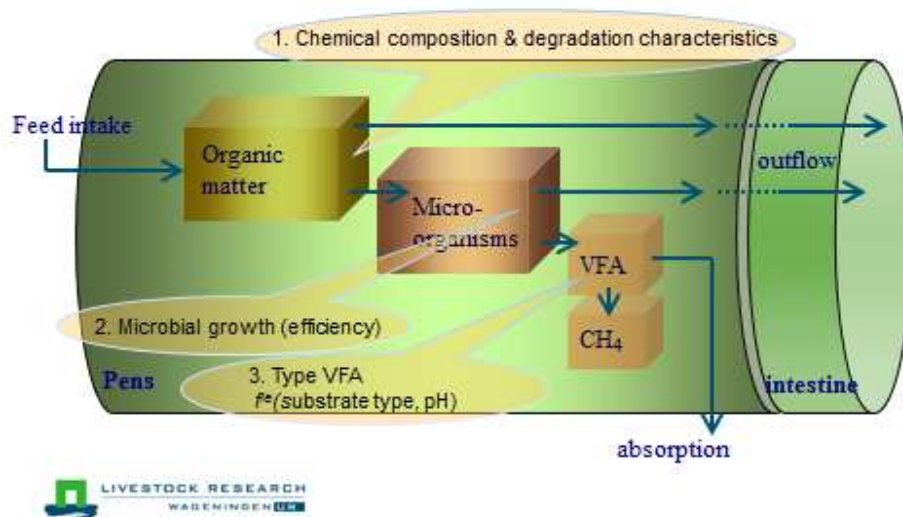
The model is driven on inputs related to nutrition, including daily dry matter (DM) intake, the chemical composition of feed DM (in principle possible to give individual and different meals as an input as well), and intrinsic degradation characteristics of the starch, crude protein and cell wall material (structural carbohydrates). Besides these degradable fractions, the model also requires input on dietary content of crude sugars, crude fat (including the degree of saturation of dietary fat), organic acids, ash and ammonia. The model predicts the process of enteric fermentation and microbial activity in the rumen and large intestine, and predicts enzymatic digestion in the small intestine by empirical equations.

In addition to feed related model inputs, the model requires some parameter values which are estimated by empirical equations already included in the model when used as a Tier 3 approach, but which can also be given as an input to the model. These parameters involve the volume of the rumen and of the large intestine, the fractional passage rates of fluid and particulate matter in rumen and the fractional passage rate of digesta in the large intestine, and three parameters (average, minimum and time period below 6.3) of daily pH dynamics in the rumen as well as for

the large intestine. Furthermore, the model contains parameters for enzymatic digestion of protein, starch and fat in the small intestine.

Finally, for prediction of milk production, the model requires protein, fat and lactose content in milk be given as an input (or simply assuming the reference values for calculation of fat and protein corrected milk).

a



b

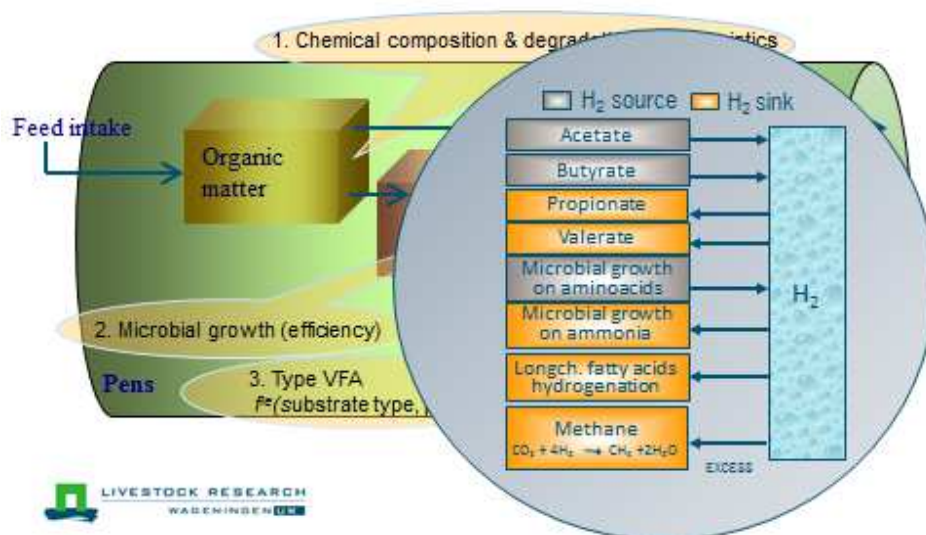


Figure 2.1. Diagram of the model structure of the Dutch Tier 3 for enteric fermentation in dairy cows, including three causal factors to explain variation in (a) microbial fermentation of feed substrate, microbial growth, production of volatile fatty acids and methane as end-products of fermentation, (b) the effect of the profile of volatile fatty acids, microbial growth and long-chain fatty acid bio-hydrogenation on hydrogen excess and methanogenesis.

2.3. Manure storage and digestion model

A process-oriented model was constructed which describes the conversions of carbon (C), nitrogen (N) and sulphur (S) in stored manure (Hutchings *et al.*, unpublished; Figure 2.2). The model requires the amount and composition of animal excreta (or manure quality), the distribution of C between fractions with a distinct degradability and a distinction between ammoniacal and organic N as an input. The model predicts emissions of CH₄, CO₂, ammonia (NH₃), N₂O, di-nitrogen (N₂), hydrogen sulphide (H₂S) from stored manure and calculates at an hourly or daily time step and can represent variation in the dynamics of the deposition of manure and manure storage time. Only a preliminary parameterization of the model has been used however, and further development is needed before any conclusive results can be shown. Besides parameterization also further attention is needed to modelling slurry temperature and the transformations taking place in the crust on top of stored manure.

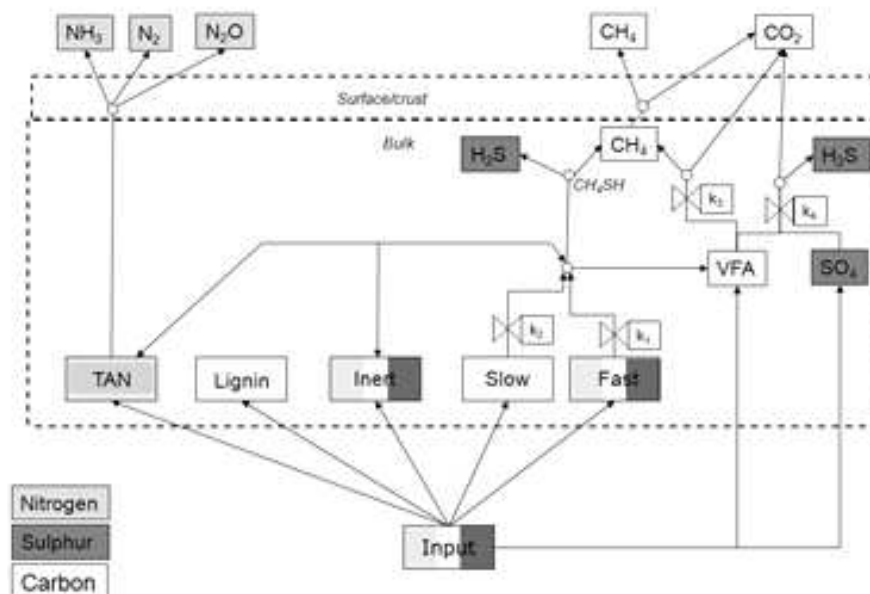


Figure 2.2. Diagram of a model of manure storage (Hutchings *et al.*, unpublished).

2.4. Pasture simulation model PaSim

2.4.1. Model representation, model aim and model use

The Pasture Simulation model was developed at INRA-UREP (PaSim, APP ID:IDDN.FR.001.220024.000.R.P.2012.000.10000; e.g. Vuichard *et al.*, 2007a, b; Graux, 2011; Graux *et al.*, 2011; Graux *et al.*, 2013; PaSim User's Guide, December 2012, https://www1.clermont.inra.fr/urep/modeles/Pasim_User_Guide-pasim_v5-3_201212.pdf) and based on a version originally provided by Riedo *et al.* (1998). It is a process-oriented grassland ecosystem model based on the Hurley Pasture Model (Thornley, 1998) whose main aim is to simulate climate change impacts on grassland services, and feedbacks of this to the atmosphere by associated GHG emissions by animals and grassland. It was first programmed in ACSL (Advanced Continuous Simulation Language) and developed at the Research Station Agroscope (Switzerland, Reckenholz) from 1997 to 2002. Since then, it is developed at the Grassland Ecosystem Research Unit of the French National Institute for Agricultural Research (France, Clermont-Ferrand). The software is now written in Fortran 90 language and contains about 60.000 lines. It is composed of submodels for plants, animals, microclimate, soil biology, soil physics and management. The 5.3 version of the model is about to be submitted at the APP (French agency for software protection).

Grassland processes are simulated on a time step of a 1/50th of a day in order to have detailed sub-daily dynamics and ensure energy budgets stability. Simulations consider a soil-vegetation-animal-atmosphere system (with state variables expressed per m²) and run over one or several years. Animal processes are simulated at pasture, excluding the barn or confined housing conditions.

As with other advanced biogeochemical models, PaSim simulates water, C and N cycling in grassland ecosystems at sub-daily time step, and was successfully tested at European conditions (Ma *et al.*, 2014). In PaSim, microclimate, soil biology and physics, vegetation, herbivores and management are interacting modules. Simulations are run at plot-scale, where animals are only considered at pasture (not during indoor periods). Photosynthetic C is either allocated dynamically to one-root and three-shoot compartments (laminae, sheaths and stems, ears) each of which consists of four age classes, or lost through animal milking, enteric CH₄ emission and returns, and through ecosystem respiration. Accumulated aboveground biomass is either cut or grazed, or enters a litter pool. Biological N₂ fixation is modelled according to Schwinnig and Parsons (1996), assuming a constant legume fraction. Vegetation is parameterized for a set of key functional traits such as the maximum specific leaf area, the light-saturated leaf photosynthetic rate in standard conditions, the fraction of fibres in ingested shoot compartments and the fraction of digestible fibres in total ingested fibres. Accumulated aboveground biomass can be utilized by cutting and grazing, or enters a litter pool.

The N cycle considers N inputs to the soil via atmospheric deposition, fertiliser addition, symbiotic fixation by legumes and animal faeces and urine. The inorganic soil N available for root uptake may be lost through leaching, volatilization and nitrification/denitrification, the latter processes leading to N₂O gas emissions to the atmosphere. Management includes mineral and/or organic (e.g. solid manure, slurry) N fertilisation, mowing and grazing, with parameters set by the user or optimized by the model according to pre-set goals.

2.4.2. Model parameterization

2.4.2.1. Climate conditions

The PaSim model was parameterized for representative grassland-livestock systems under conditions represented by 12 sites in France (Figure 2.3). Exemplary simulations are given for basic mitigation options at four sites, which cover contrasting agro-ecological zones (Table 2.1). Three contrasting years in terms of aridity (humid, median and arid) were selected over 1970-2006 at each site (observed climate data, Table 2.1) according to the De Martonne-Gottmann aridity index ($0 \leq b < \infty$ [extreme humidity]).

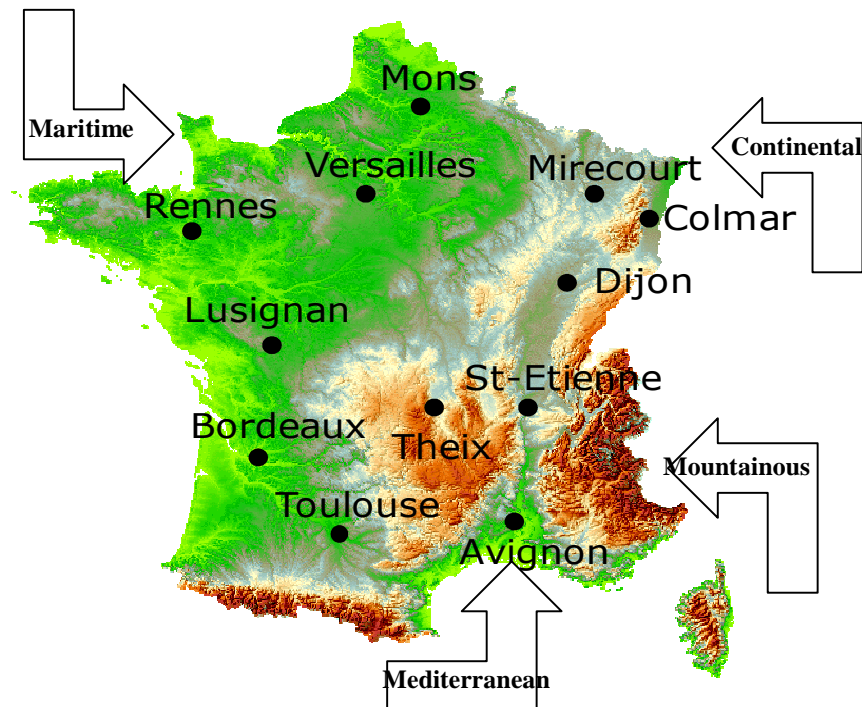


Figure 2.3. Location of 12 French study sites (from the ANR CLIMATOR project, http://www.international.inra.fr/research/green_book_of_the_climator_project). Four sites were selected to represent contrasting agro-ecological zones.

2.4.2.1. Soil conditions

The PaSim model was initialized with soil organic matter values (SOM) obtained by running spin-up simulations until equilibrium was reached. To test the three mitigation options, simulations were run on 0.8-1.0 m depth limestone brown soil. For the first two options, two scenarios were configured with low or high initial soil organic matter (SOM) content.

Table 2.1. Geo-location and climate type of sites presented in this study. Climate types were classified according to three complementary indicators: continentality (Emberger, 1930), Mediterraneity (Le Houérou, 2004) and aridity (De Martonne, 1942). Mean air temperatures and rainfall totals are reported for the period of available years.

Site	Latitude	Longitude	Altitude (m a.s.l.)	Climate type	Rainfall (mm yr ⁻¹)	T _{avg} (°C)	Years
Avignon	43° 54' N	04° 54' E	37	Sub-Mediterranean, semi-arid to arid	702	14.0	1970-2006
Mirecourt	48° 18' N	06° 08' E	265	Semi-continental, humid to sub-humid	877	9.2	1973-2006
Rennes	48° 06' N	01° 42' W	35	Lowland littoral, sub-humid to semi-arid	727	11.4	1975-2006
Theix	45° 43' N	02° 08' E	890	Mountain, humid to sub-humid	774	7.9	1971-2006

Table 2.2. Selected contrasting years in terms of aridity, based on the De Martonne-Gottmann aridity index (*b*).

Site / Aridity conditions	Humid		Median		Arid	
	Year	<i>B</i>	Year	<i>b</i>	Year	<i>b</i>
Avignon	1996	27.1	2000	14.8	1989	6.3
Rennes	1994	26.3	1977	18.6	1989	11.9
Theix	1979	37.3	1998	25.5	1985	13.8
Mirecourt	1999	45.0	1979	28,2	2003	14.9

2.5. Soil denitrification-decomposition model DNDC

2.5.1. Model representation, model aim and model use

For the work presented in the present study the version 9.5 of the DNDC model was used. The User's Guide for the DNDC Model (Version 9.5) of August 2012 provides the following extensive description of DNDC.

The DeNitrification-DeComposition (DNDC) model is a process-oriented computer simulation model of carbon and N biogeochemistry in agro-ecosystems. The model consists of two components. The first component, consisting of the soil climate, crop growth and decomposition sub-models, predicts soil temperature, moisture, pH, redox potential (Eh) and substrate concentration profiles driven by ecological drivers (*e.g.*, climate, soil, vegetation and anthropogenic activity). The second component, consisting of the nitrification, denitrification and fermentation sub-models, predicts emissions of CO₂, CH₄, ammonia (NH₃), nitric oxide (NO), N₂O and dinitrogen (N₂) from the plant-soil systems. Classical laws of physics, chemistry and biology, as well as empirical equations generated from laboratory studies, have been incorporated in the model to parameterize each specific geochemical or biochemical reaction.

The entire model forms a bridge between the C and N biogeochemical cycles and the primary ecological drivers (Figure 2.4).

Plant growth plays an important role in regulating the soil C, N and water regimes, which could further affect a series of biochemical or geochemical processes occurring in the soil. A sub-model was built in DNDC to simulate the crop growth. A group of crop parameters can be provided or modified by the users to define their own crop. The crop parameters include maximum yield, biomass partitioning, C/N ratio, season accumulative temperature, water demand, and N fixation capacity. The crop growth will be simulated driven by the accumulative temperature, N uptake, and water stress at a daily time step. The modelled daily photosynthesis, respiration, C allocation, and water and N uptake are recorded so that the users can check the modelled results against their observations to make sure the crops are simulated correctly. All the crop parameters are accessible on the user's input interface so that the users can modify the parameters in a prompt mode. Crop demand for N is calculated based on the optimum daily crop growth and the plant C/N ratio. The actual N uptake by crop could be limited by N or water availability during the growing season. After harvest, all the root biomass is left in the soil profile, and a user-defined fraction of the above-ground crop residue remains as stubble in the field until next tilling application, which incorporates the stubble onto (for no-till) or into (for conventional tillage) the soil profile. The crop residue incorporated in the soil will be partitioned into three soil litter pools, namely very labile, labile and resistant litter pools, based on its C/N ratio. The litter incorporation provides essential input for the soil organic matter (SOM) storage and hence integrates the plant and soil into a biogeochemical system.

In DNDC, SOM resides in four major pools: plant residue (*i.e.* litter), microbial biomass, humads (*i.e.* active humus), and passive humus. Each pool consists of two or three sub-pools with different specific decomposition rates. Daily decomposition rate for each sub-pool is regulated by the pool size, the specific decomposition rate, soil clay content, N availability, soil temperature, and soil moisture. When SOC in a pool decomposes, the decomposed carbon is partially lost as CO₂ with the rest allocated into other SOC pools. Dissolved organic carbon (DOC) is produced as an intermediate during decomposition, and can be immediately consumed by the soil microbes. During the processes of SOC decomposition, the decomposed organic N partially transfers to the next organic matter pool and is partially mineralized to ammonium (NH₄⁺). The free NH₄⁺ concentration is in equilibrium with both the clay-adsorbed NH₄⁺ and the dissolved ammonia (NH₃). Volatilization of NH₃ to the atmosphere is controlled by NH₃ concentration in the soil liquid phase and subject to soil environmental factors (*e.g.* temperature, moisture, and pH). When a rainfall occurs, NO₃⁻ is leached into deeper layers with the soil drainage flow. A simple kinetic scheme "anaerobic balloon" in the model predicts the soil aeration status by calculating oxygen or other oxidants content in the soil profile. Based on the predicted redox potential, the soil in each layer is divided into aerobic and anaerobic parts where nitrification and denitrification occur, respectively. When the anaerobic balloon swells, more substrates (*e.g.* DOC, NH₄⁺, and N oxides) will be allocated to the anaerobic microsites to enhance denitrification.

When the anaerobic balloon shrinks, nitrification will be enhanced due to the reallocation of the substrates into the aerobic microsites. Gases NO and N₂O produced in either nitrification or denitrification are subject to further transformation during their diffusion through the soil matrix. Long-term (*e.g.* several days to months) submergence will activate fermentation, which produces hydrogen sulfide (H₂S) and CH₄ driven by a decrease of the soil Eh.

2.5.2. Model inputs and outputs

The entire model is driven by four primary ecological drivers, namely climate (precipitation, wind speed, sun hours/radiation, humidity, temperature), soil (water tables, pore size, soil type, soil OM fractions), vegetation (cropping, pasture, crop residues, rooting), and management

practices (tillage, manure application, artificial fertiliser application, soil structure). It is inherently important for a successful simulation to obtain adequate and accurate input data about the four primary drivers. Realistic input based on farm case specific monitoring (e.g. fertilisation, response of vegetation, soil and pasture management) is required to obtain realistic model outcomes and to explore effects of mitigation in a realistic manner. The model predicts emissions of N_2O , CO_2 and (soil) CH_4 in relation to the predicted responses of vegetation, SOM, and soil nitrification/denitrification processes.

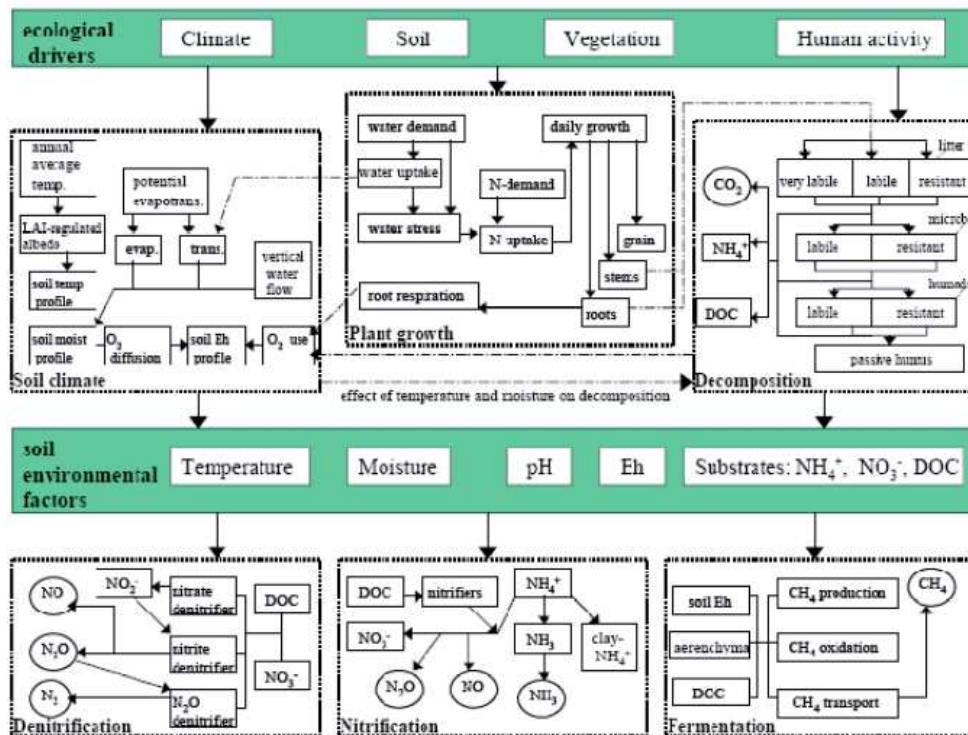


Figure 2.4. Diagram of model structure of DNDC, version 9.5 (Li et al., 2011).

2.5.3. Model farms used in DNDC

Two research farms were used for the DNDC calibrations. Johnstown Castle is located in the South-eastern corner of Ireland, an area typically characterised by a large percentage of tillage activity due to the free-draining soils and relatively drier climate. Solohead is located in the southern midlands, which is the principal dairy producing area in Ireland (see Figure 2.5). Johnstown dairy farm is a grazed dairy system, on ryegrass-predominated pastures with stocking rates of 2.9 LSU ha^{-1} and between $180 - 230 \text{ kg N ha}^{-1}$ applied annually. Soils are eutric cambisols and are moderate to free-draining. Solohead has lower stocking rates (2.2 LSU ha^{-1}) with $60 - 220 \text{ kg N ha}^{-1}$ applied annually. The large variation in N application rate is due to the fact that half of the farm had 20% ryegrass/clover swards.

Table 2.3. Site characteristics of Johnstown Castle and Solohead Farms

Site	Latitude	Longitude	Soil type	SOC	Rainfall	T _{avg}	Years
				(tC ha ⁻¹)	(mm yr ⁻¹)	(°C)	
Johnstown	52.29N	6.50 W	Eutric Cambisol	121	1102	9.9	1980-2012
Solohead	52.50N	8.21 W	Gleysol	155	1312	9.4	1980-2012



Figure 2.5. Map of Ireland showing the location of Johnstown Castle and Solohead Farm.

3. Mechanisms of trade-offs between GHG emissions with mitigation measures at animal, manure, field and soil level

This chapter discusses the mechanisms involved with the on-farm GHG emission from various farm elements (*i.e.* animals, manure storage, field and soils). The mechanisms responsible for these GHG emissions have been represented by process-based models (described in Chapter 2). These models are able to quantify in detail the consequences of mitigation measures on CH₄, N₂O and CO₂ emissions (and C sequestration in soils). Various GHG mitigation measures have been studied and quantification of the consequences for these GHG emissions was documented in the previous deliverables D8.2 and D8.3 (first and second version of process-oriented estimates of mitigation and adaptation options). This chapter focuses on quantification of the trade-offs between GHG emissions from the various farm systems elements, and will be discussed from the perspective that for understanding these trade-offs the mechanisms underlying them must be understood, and quantified. For the mitigation measures documented previously it will be discussed what aspects or functionality of insights obtained with these process-based models needs to be taken into account when quantifying the trade-offs between different sources of GHG. It will be discussed how to use such insights when attempting to address these trade-offs in farm system GHG inventories, and to be able to account for them.

3.1. Reducing nitrogen fertilisation rate

Reducing N fertilisation rate affects the characteristics and nutritional value of forages. This particularly holds for grasslands or grass products, and less for the characteristics and nutritional value of crops such as wheat and maize. Fertilization rate does affect yields of all forages, but it is expected to affect less the eventual characteristics of the products harvested from wheat and maize. An important side-effect of lower N fertilization rate to be taken into account is the consequence of lower crop yields on dietary composition. If lower crop (*e.g.* maize) yield as a result of lower N fertilization affects a higher inclusion rate of grass products in the diet this will with a high certainty lead to higher enteric CH₄ emission per unit of animal product (Bannink *et al.* 2014). Conversely, a lower N fertilization rate of grassland may lower grass yield and hence lead to supplementation of the diet with other crop harvests. In this case the effects on enteric CH₄ are not obvious because a lower N fertilisation rate leads to increased enteric CH₄ from grass which at least partly compensates the N₂O sparing effect. Supplementation of the diet leads to another diet composition and likely to a reduction in enteric CH₄ emissions per unit of animal product again, in synergy with the lower N₂O emissions from grasslands. However, including other feeds, feed crops or by-products implicates that part of the GHG emissions associated to their production is put outside the physical farm boundaries.

Models for farm GHG inventory can easily take into account shifts in dietary composition and determine the likely consequences for animal productivity. However, process-based models are better suited to quantify in detail the effects on enteric CH₄ and the consequences for the amount, site (urine vs. faeces) and form (type of N compound) of N excreted. These details are important when estimates are to be made of the consequences of lower N fertilization rate on enteric CH₄, N excretion, N emissions and related (direct and indirect) N₂O emissions.

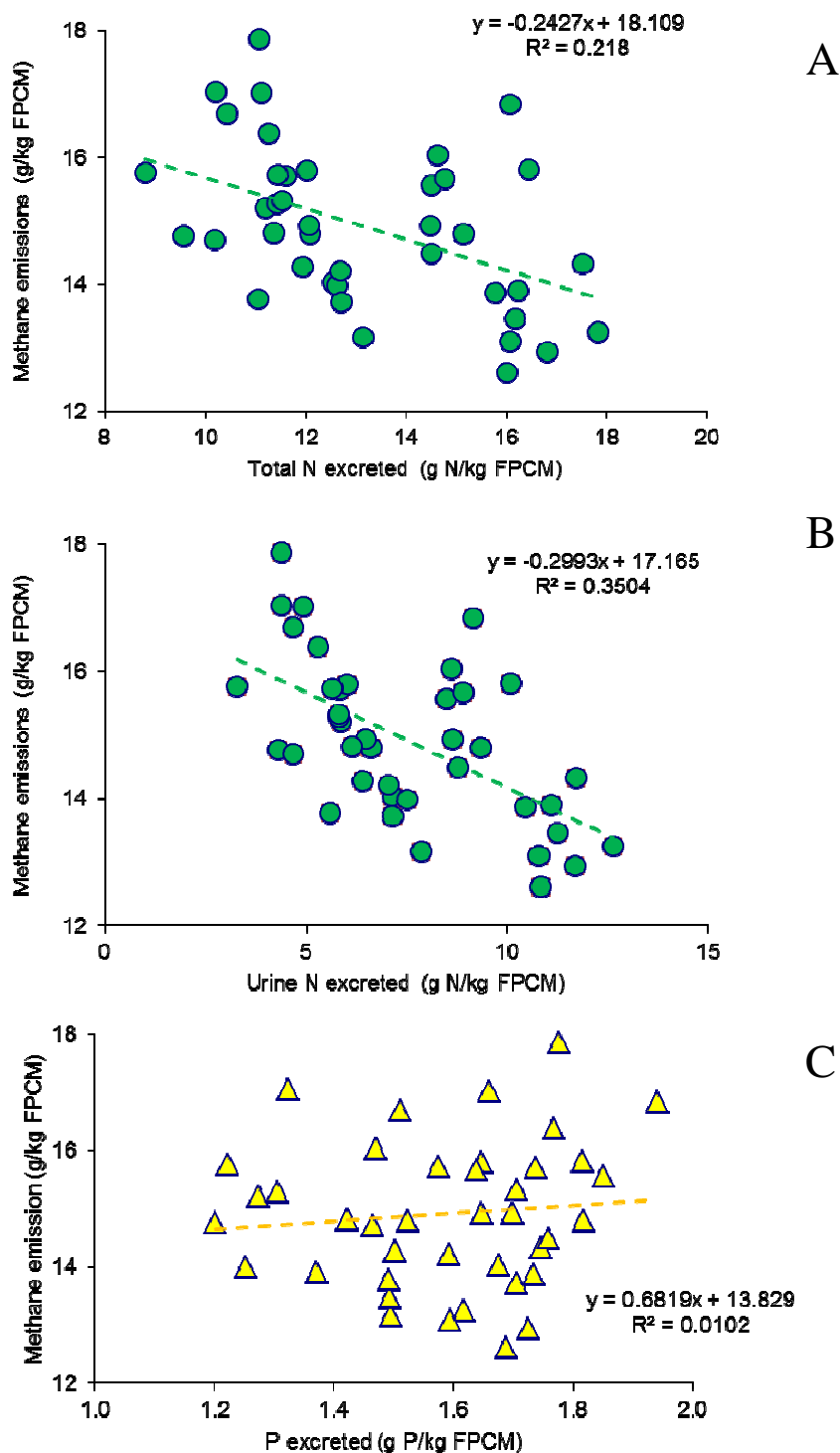


Figure 3.1. Simulation results with the Dutch Tier 3 model for the effect of N-reducing nutritional changes in dairy diets on enteric CH₄ emission, total N excretion (A), urine N excretion (B) and P excretion (C), obtained with a predecessor of the Dutch Tier 3 model (Reijs, 2007; Dijkstra et al. 2011 & 2013; Bannink et al., 2013b).

A previous simulation study on the effects of N excretion reducing nutritional measures in dairy farming, or vice versa, the effect of relying more on grass silage, demonstrates the effects on enteric CH₄, on the amount and form of N excreted and on P excreted (Reijs, 2007; Dijkstra *et al.* 2011 & 2013; Figure 3.1). Even though large variation remains, and the results strongly depend on the type of nutritional measure taken, a general trend was simulated of increased enteric CH₄ emission per kg of milk with a reduction in N excreted per kg of milk (Figure 3.1A). A better relationship was obtained for urine N excreted (Figure 3.1B). There was no relation between the effects of the N reducing nutritional measures on enteric CH₄ emission and P excretion (Figure 3.1C). The latter results indicate that a trade-off between GHG emissions should focus on diet composition, diet digestibility and the consequences for production. Even though P excretion is an important aspect in farm management, it is the result of the balance between P ingested and P retained in animal product, with P almost completely digestible. In contrast to the digestion, utilisation and excretion of N, that of P is rather unrelated to the process of enteric fermentation (as long as P allowance is not limiting). Replacing grass silage with N-poor forages or N-poor by-products as an N emission reducing nutritional measure, on average, increased enteric CH₄, largely compensating the effects on reduced N emissions from excreta. This general trend indicates at least the importance of taking into account the trade-off between enteric CH₄ and N excretion (NH₃ and N₂O emission). The methodology of farm inventory should accommodate for this trade-off. In previous farm surveys (Schils *et al.*, 2007) it was for example concluded that N surplus per hectare mainly drives farm GHG emissions. However, the calculations did not include the trade-off with enteric CH₄ of N surplus is reduced. The current results indicate that an increase in enteric CH₄ may easily compensate the reduction achieved in N₂O emissions with reduced N surplus if this accompanied with large changes in grass characteristics. The relationships in Figure 3.1 should not be used to quantify this trade-off between CH₄ and N₂O as they only reflect the overall effect for N mitigation nutritional measures tested and cannot be considered the outcome of study of a specific farm case. Notably, the relationships indicated in Figure 3.1 should hence not be applied to specific farming conditions or diets, or in farm survey methodology as a manner to quantify this important trade-off.

The large variation in results indicates that choices in the type of N reducing nutritional measure will affect not only N₂O emission but CH₄ emission as well even though diet digestibility and animal productivity might not seem that different. Furthermore, the large variation that remains unexplained (> 60%; Figure 3B) indicates that a detailed analysis is needed to become conclusive about the size of these trade-offs.

In correspondence with this, the results for N lowering nutritional measures tested with D8.3 (Bannink *et al.*, 2014) also indicated that the choice of feed supplemented to a grass-based diet affects CH₄ emission. A standard concentrate mixture (low cost concentrate without a high inclusion of starch-rich products such as wheat, and without protein-rich products such as soybean meal) as well as molasses delivered higher levels of enteric CH₄ per unit of milk compared to starch-rich products such as wheat, maize or maize silage.

<regressies niet van toepassing op farm system level; no predictive purposes >

Because most ruminant diets are mainly composed of grass products, next to changes in diet formulation caused by changes in grass and (forage) crop yields following a reduction in N fertilisation rate, an aspect of main importance is the effect of a reduced N fertilisation rate on the grass characteristics. In D8.3 simulation results were shown which indicate a rather large effect of N fertilisation and stage of maturity of the grass on enteric CH₄ emissions with a difference of about 25% in enteric CH₄ per kg of milk produced (similar feed intake assumed here). Such a large difference with a reduction in N fertilisation rate and in ingesting/grazing or cutting the grass in a later stage of maturity are the outcome of a changed grass digestibility and a changed chemical composition. These results are in accordance with previous modelling

results which indicated an even larger range of CH₄ emission per unit of milk (Bannink *et al.*, 2010). The result that N fertilisation rate and related changes in grassland management can affect CH₄ emission from grass was recently confirmed by in vivo measurement for grass herbage fed dairy cows (Figure 3.2; Warner *et al.*, 2015). The authors concluded that altering grass quality through an increase of N fertilization and a shorter regrowth interval reduced CH₄ emission. The larger amount of CH₄ produced per day and cow with the more intensively managed farms was compensated by a higher feed digestibility and higher milk yield. The CH₄ emission in g per kg of fat and protein corrected milk ranged from 14.6 (observed for the high N fertilisation and early cutting) to 17.4 (observed for the low N fertilisation and late cutting), which is about 20% difference and in size very similar to what is predicted by the Dutch Tier 3 model (Bannink *et al.*, 2014; D8.3). When CH₄ is expressed in terms of % of gross energy ingested by the cows, following the IPCC Tier 2 methodology, the value ranged from 6.4 to 7.1% for low and high N fertilisation rate, resp., which is about half the size of difference obtained for g CH₄ per kg milk. Meanwhile observed N excretion ranged in a reverse manner from 195 to 138 g N excreted per day. There were no interactions for effect of N fertilisation rate and stage of grass maturity. The observations by Warner *et al.* (2015) correspond well to earlier simulations with the Dutch Tier 3 model for grass herbage diets by Bannink *et al.* (2010) which indicated an increase from 6.3% to 7.0% with a reduction of N fertilisation rate. On average, observed values by Warner *et al.* (2015) and simulated values by Bannink *et al.* (2010) exactly match and indicate 6.7% of gross energy intake is emitted as CH₄ from grass herbage. This value is very close to the IPCC Tier 2 adoption of 6.5%, and slightly higher than the 6.3% predicted for. An important finding of Warner *et al.* (2015) was that it depended on the unit in which CH₄ was expressed whether a significant effect could be established. This outcome reflects how the underlying mechanism of CH₄ emission responds to the changed grass herbage characteristics with a major effect of stage of maturity on digestibility (reflected in CH₄ per kg dry matter intake) and the different value of the grass for the cows (reflected in CH₄ per kg fat and protein corrected milk produced). These in vivo results hence confirm the possibility that grassland management strongly impacts enteric CH₄ emission and in a reverse manner N excretion contributing to N₂O emissions.

When not specifically addressed in farm monitoring and GHG inventories the effects of these trade-offs remain unnoticed. However, based on the insights obtained with process-based models and the confirmation by empirical data, we know that these trade-offs do exist. This means that for realistic results of inventories on the effect of mitigation measures under adaptation to climate change, these trade-offs have to be accounted for in the methodology adopted. Methodologies for surveys or inventories evaluating grassland management and N emissions have to accommodate the most important effects as a result of the mechanisms underlying these trade-offs (dietary and digestive aspects of ruminant nutrition, soil processes and yields of grass and crops, processes related to emissions from excreta, manure storage and soil fertilisation), not only on a farm scale but also on a regional and national scale. The process-based models clearly are needed to aid in finding routes for 1) how to identify such mechanisms underlying trade-offs, and 2) how to apply them in a case-specific manner in order to be able to link them to farm management options and strategies, and 3) how to make them applicable by adjustment of GHG monitoring and inventory methodology.

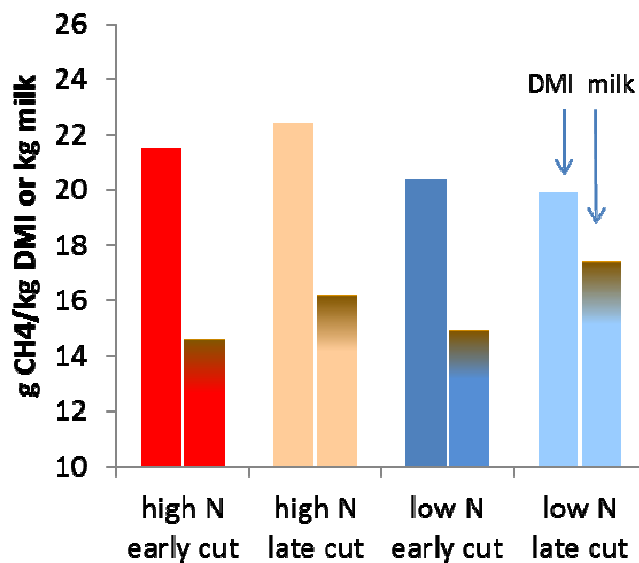


Figure 3.2. *In vivo* observations (Warner *et al.*, 2015) of the effect of N fertilisation rate (high N, 90 kg of artificial fertiliser N following initial cut; low N, 20 kg N following initial cut) and stage of grass maturity (early cut, 3 weeks of regrowth after initial cut; late cut, 5 weeks of regrowth after initial cut) on enteric CH₄ emission expressed in g kg⁻¹ dry matter intake (left bars; DMI) or in g kg⁻¹ fat and protein corrected milk (right bars; milk). Stage of maturity or weeks of regrowth had a significant effect on CH₄ per kg corrected milk ($P=0.015$) and N fertilisation rate had a significant effect on CH₄ per kg dry matter intake ($P=0.017$).

The fore-mentioned effects of reducing N fertilisation rate of crops or grassland or of the combination of both, may have a strong impact on enteric CH₄ emission by changed characteristics of the grass itself, but also by changed feeding practices because yield of grass and crops may be affected (particularly when considered in combination with the higher vulnerability for more extreme weather conditions with climate change). Current methodologies for inventory of farm GHG emissions generally limit themselves to represent the effect on available animal feed and feed digestibility. The important aspect of feed characteristics and feed quality is addressed in an only very limited manner which is not enough to be able to account for the effects it might have on emission of enteric CH₄ and N₂O from excreta and soils. Enteric CH₄ is calculated as a direct function of intake and diet digestibility (comparable to the IPCC Tier 2 approach for enteric CH₄). However, if digestibility is not affected such methodology would also predict no effect on enteric CH₄, which is in clear contrast with the results from the Dutch Tier 3 model documented in D8.3 (Bannink *et al.*, 2014) and the results of experiments (*e.g.* Warner *et al.*, 2015). Those results clearly identify that important trade-offs have to be expected that could (partly) offset the beneficial effect of reducing N fertilization rates.

In conclusion, a main focus of identifying the mechanisms involved in the trade-offs should lie on the relationship between the reduced N fertilization rate and the lower soil N emissions on the one hand, and the effect on animal production and enteric CH₄ emissions on the other hand. Farm inventory methodologies which do not address the complexity and process details as process-based models do, cannot bring view into how such trade-offs could take place and are not suitable to quantify them without any particular arrangements in these models that would allow them to do so.

3.2. Introduction of legumes, or permanent / multispecies grasslands

Farm strategies are needed to make grassland more resilient to extreme weather, in particular in Southern Europe where draughts may be occurring more frequently and more severely under the more extreme climate change scenarios. A potential adaptation strategy is the introduction of legumes in grass swards, or the introduction of more permanent multispecies grassland. Long-term draughts seriously deteriorate grassland-based systems not only since grass as a feed would be lacking but also since grass C stocks in soil could be degraded which would lead to high CO₂ emissions. Considering the size of the soil C stocks involved with grasslands, this is a serious trade-off of not adapting towards less climate sensitive and more climate-resilient grassland systems. A total deterioration of grassland and full degradation of soil C stocks would have long-term effects, including loss of soil fertility, that cannot be met in the short term by adapted farm management.

Although such legumes-rich, or multi-species/permanent grasslands may improve resilience against extreme weather conditions and in this manner ensure existence of a mainly grass-based livestock production system, there is also the potential trade-off of higher N₂O emissions rates. In D8.3 it was simulated with both DNDC and PaSim that introduction of up to 30% legumes delivers hardly any trade-off with respect to N₂O emission whereas legumes may contribute to N availability in soil and improve N utilisation by grass species in combination with legumes species. Above this 30%, N₂O emission starts to rise steeply however and this trade-off may counteract the improved N utilisation (at low N fertilisation rates) by grass / legumes mixtures. Furthermore, this trade-off occurred for farming systems under conditions ranging from humid to arid.

Another trade-off of introducing legumes and multispecies grassland is the associated potential for lower yields and lower feed qualities. This will reduce feed intake and animal productivity and may hence strongly inflate GHG emission density when expressed per unit of animal product obtained. Also, lower N fertilisation rates and lower animal stocking density will probably reduce C sequestration in grassland which is a trade-off which needs to be taken into account with this adaptation option towards more resilient grass swards. These aspects have been simulated with the process based models PaSim and DNDC and results were presented in deliverable D8.3. The outcome was that N fertilisation rate and animal stocking density can have a profound impact on N₂O, CH₄ and CO₂ emissions. Increasing the legumes fraction or shifting towards more resilient multispecies grasslands was on a net basis not very effective in reducing GHG emissions, however yields were simulated to be less sensitive to N fertiliser inputs.

Depending on the stocking density, soil conditions, and grass growth conditions, introduction of legumes must be expected to have varying success. In the more temperate regions of Europe with relatively high N fertilisation rates it may prove to be difficult to maintain a substantial proportion of legumes in the grass sward without intensive management and require continuous reseeded as described for the grass/clover system on the Irish experimental farm discussed in D8.3. From a nutritional perspective, there are no reasons that legumes cannot deliver a good quality forage for ruminants with a high nutritive value. No specific simulation studies have been performed in this respect with the process-based model because the detailed information needed as model inputs is essentially lacking. Nevertheless, the protein fraction in legumes is generally known to be more digestible than in grass species. Although this is rather a disadvantage in intensively managed grassland systems that may lead to relatively high N losses thereby increasing N₂O emissions, this may be an advantage in extensively managed grassland systems in more arid conditions where legumes may overcome the potential shortage of digestible N in grass as a result of the relatively high proportion of indigestible N under those conditions. In such cases a higher legumes proportion may prevent the need for supplemental

protein feeding or N inputs which also has a sparing effect on GHG emission associated with purchase of the external inputs.

Whether a higher legumes proportion is feasible and fits into the management strategy of livestock farms will depend on the level of intensity of that farm however. Low input, multispecies grassland is not likely capable of keeping productivity levels up to the current level of intensive systems. A reduction of grass yield and growth rates due to a higher proportion of legumes and more species-rich grass sward must be expected which will lower animal productivity as well. This is a profound trade-off towards GHG emission intensity per unit of animal product. A comparison of simulation results obtained with the process-based models between a no grazing, a partial grazing and an unrestricted grazing dairy production system, discussed in D8.3, clearly demonstrated the trade-offs between the intensity of the farming system and intensity of GHG emission per unit of animal product. The most intensive system did not show much higher on-farm GHG emissions, however this result excluded still the GHG emissions external to the farm. Although inclusion of these external GHG emission would likely made GHG emission per unit of milk in the intensive system higher than for the low input, partial grazing farm and the unrestricted grazing farm, the example nicely demonstrates that accuracy is needed in quantifying separate GHG sources and sinks in order to become conclusive about the trade-offs of shifts in farm management like the introduction of high proportions of legumes in grassland.

In adapting to climate change, grazing-based systems might have to adapt to temporal shortages in feed supply. An adaptation strategy could be to introduce feed stocks. Such an adaptation strategy involves a systemic change of the farming system however, which will have profound effects on feed quality, animal productivity and associated GHG emissions. Addressing the changes in GHG emissions and evaluating the adaptation capacity of farming systems requires that a link is made in quantitative terms between the implications of management options and GHG emissions, including assessment of the trade-offs that exist.

In conclusion, although there seems good potential for the introduction of legumes and multispecies grass swards as an adaptation to climate change in the most vulnerable areas, it is unclear what the precise consequences systemic changes will be. Information is scarce on how climate change will affect the grass sward characteristics and what the consequences would be on enteric CH₄ emission, nutritive value, livestock productivity and as a result vulnerability of N emission from excreta. Such information is pivotal in reliably estimating trade-offs between GHG emissions when introducing (a higher proportion of) legumes in grasslands. More detailed information about these consequences also requires accountability of these effects by methodologies used for farm system monitoring or comparison, and methodologies used for GHG inventories used for evaluating policy scenarios and in socio-economic surveys.

3.3. Exchange between grassland production and forage crop production

Climate change may force adaptation of ruminant production systems relying mainly on grazing in the direction of applying feed stocks, or even a shift from grassland to forage crop production. Such a change implies a major impact on the farming system and may have major complications for soil C stocks and soil fertility in general. A possibility to adapt may be a shift towards crops whose nutritive value is less vulnerable for warm and dry conditions preceding harvest, such as cereal grains and maize. Although such seasonal conditions seriously affect grass sward productivity, grass nutritive value, animal productivity and enteric CH₄ emission from grass, harvesting grains and maize may provide for making stocks of feed to overcome the seasonal shortage. Although feed stocks can also be made from grass (hay, silage produced

before and after the season with poor productivity), their nutritive value will be more negatively affected during than what can be expected for maize.

From the viewpoint of reducing enteric CH₄ emission the exchange of grass products for other starch-rich forage crops is highly feasible. Some 10 to 15% less enteric CH₄ is formed with maize silage compared to grass silage. However important trade-offs exist with soil CO₂ emissions when converting grassland into crop production, which compensate the mitigation of enteric CH₄. Furthermore, there might be a compensatory effect towards CH₄ emissions from manure. Changing from grazing systems to more confined systems of dairy production also introduces manure stocks in the farm management which in themselves become large sources of CH₄ emissions, in particular when manure is stored under warm conditions. In addition there can be a nutritional set-off towards CH₄ emission from manure because of a higher amount of indigested C ending up in manure where it forms a source of CH₄ emissions. Not only the amount of manure C, also manure storage conditions and the degradability of manure C determines the amount of CH₄ emitting.

These aspects have been studied with the process-based manure model described in deliverable D8.3. Simulations indicated for example a decrease in manure CH₄ emission with a decreased proportion of grass silage in exchange with maize silage as a result of a far lower degradability of manure C originating from maize fibre compared to that from grass silage fibre. Furthermore, comparing different grassland management options indicated that the largest CH₄ emission from manure is expected for grass silage under a high N fertilisation and early cutting regime, and the lowest emission for the low N fertilisation and late cutting regime. A substantial 40% difference in manure CH₄ emission rate was estimated for both regimes, and when realizing that 25 to 30% of CH₄ emissions may originate from manure instead of enteric fermentation, such differences have a profound impact on the farm GHG budget that cannot be ignored but must be taken into account as a potentially important trade-off of nutritional measures to reduce N excretion or enteric CH₄ emission. Effects can also be synergistic instead of being a trade-off. For example, less enteric CH₄ because of the exchange of grass for maize may also allow more control on N excretion, reducing N emissions as well. Furthermore, inclusion of increasing proportions of maize silage delivered less degradable manure C and reduced manure CH₄ emissions next to the mitigating effect on enteric CH₄ emissions. Such results need to be confirmed by observations however and monitoring of farm cases. Little is known about the consequences for production and GHG emission intensity if such profound changes in type of forage produced would be implemented in managing the continuity of feed supplies and managing farm land in areas where farms are most vulnerable to climate change, seasons with harsh conditions and temporarily extreme arid conditions. Although process-based models can support in delineating the impact of such changes on enteric CH₄, N₂O and CO₂ emissions from crops and soils, and CH₄ emission from manure, the management options chosen, including management of feed stock but also managing livestock (e.g. prevention of heat stress), have such a profound impact on whole farm activity that collection of activity data for farms under adaptation to climate change deserves the highest priority.

An evaluation of systemic changes brought about in farming systems under climate change and of the consequences of the management options available to farmers will have such a large impact on GHG emissions and sustainability of these systems, that main focus should be on these management aspects. Before accuracy of estimates of individual GHG sources and their trade-offs can be obtained, obtaining reliable and realistic activity data is the biggest challenge and monitoring such activity is probably the first requirement to investigate management options for farmers to adapt to climate change. Subsequently, the potentials for mitigating GHG emission can be explored, also making use of process-based models to identify emission factors and trade-offs.

3.4. Mitigating by change of diet supplementation

In more intensive ruminant production systems, choices can be made for supplementing the diet with concentrates and by-products. In deliverable D8.3 simulation results were discussed for some supplement alternatives which indicated that supplement choice must be expected to affect enteric CH₄ as well. Grass herbage (high N) and untreated soybean meal showed the largest CH₄ emission per kg of milk produced, compared to the other supplementations tested (treated soybean meal, low N grass silage with or without urea). Also the choice of carbohydrate supplementation was tested and demonstrated higher CH₄ emissions per kg of milk with molasses and standard concentrates compared to starch-rich supplements wheat, maize or maize silage (deliverable D8.3).

Although certainly not meant to be decisive for what supplement best be added to mitigate enteric CH₄ and simultaneously optimize the dairy diet, the simulation results did indicate the sensitivity of enteric CH₄ emission for the choices made for protein and carbohydrate/energy supplementation of dairy diets with emission intensities typically ranging by some 15% for both types of supplementation. This outcome infers that methodologies to evaluate GHG emissions should be able to accommodate such variation. Differences simulated with the process-based model are large enough to warrant accountability of these nutritional details. A methodology limited to the concept of energy requirement and diet digestibility, such as the IPCC Tier 2 approach, is incapable to address such details. If nutritional measures (energy and protein supplementation) are to be evaluated in GHG inventories, some important aspects of enteric fermentation contributing to CH₄ formation have to become represented in these models. As demonstrated in D8.3, this holds both for intensive farming systems and for extensive farming systems with supplemental feeding.

3.5. Application of N₂O mitigating compounds

Simulated effects of nitrification inhibitors on N₂O emission with the process-based model DNDC were documented in deliverable D8.3. Although there was quite a discrepancy between modelled and observed effects of the inhibitor on N₂O and N₂ emissions, the model did reproduce that substantial reductions can be achieved with this inhibitor, and it reproduced the difference in emission rates for two different locations realistically (*i.e.* experimental farms of Johnston Castle and Solohead, Ireland).

Although clearly further modelling work is needed to improve prediction of the efficacy of inhibitors to mitigate N₂O emission, these results do demonstrate the value of the DNDC model to identify the mechanism underlying such reductions. Another value of applying process-based models is that they are not based on empirical data as heavily as higher Tier approaches are. For example, just as the process-based model could be used to evaluate the consequences for manure production and composition and the concurrent GHG emission from manure (D8.3), this process-based model could just as well be used in combination with the DNDC model to evaluate the effectiveness of an inhibitor excreted together with urine N. Urine N can be the origin of massive N₂O emission in the field, in particular when animals group together or when animals urinate repeatedly on the same location, leading to extreme N loading of soil. Current modelling results demonstrate that the process-based models may help to improve methodologies to survey impacts of farm management on N excretion in the field and related N₂O emissions. Farms adapting to climate change may have profound alterations of animal behaviour and performance (*e.g.* shading during the day in the warm season, feed intake, fate of excreta N) which will impact the site and intensity of N₂O emissions. Process-based models may prove to be very useful to delineate how conditions will impact these emissions, in order to have

them accounted for in inventory methodologies and farm surveys. This may also involve the delineation of the mechanisms that determine efficacy of soil or animal applied nitrification inhibitors, and the extent to which changes in farm management due to adaptation to climate change and the trade-off with respect to field N₂O emission can be offset by introduction of these inhibitors (e.g. addition to site where supplemental feed and high N excretions coincide).

3.6. Alternative storage / processing of animal manure

Whereas manure management does hardly contribute to the control of GHG emissions with unrestricted grazing and pasture-based livestock production, various technological options are thinkable to handle manure in the more confined systems with a high intensity of feeding and manure management. Both N and C emission processes from manure are highly temperature dependant which makes that climate change may impact on NH₃ emissions from excreta and on NH₃ and CH₄ emission from stored manure. A direct trade-off can easily occur for reducing N emissions from excreta/manure on the one hand and N₂O emission from the remaining soil-applied manure N on the other hand. This trade-off can easily be quantified by N balance calculations and accounting for the gaseous N losses before manure application by use of standard emission factors. However, the N emissions from excreta and manure in stalls and soils are far more determined by conditions and farm management (temperature, volumes, application rates and application timing, soils conditions, crop requirements) than by N balance calculation.

Accounting for trade-offs between (direct and indirect) GHG emissions from animal manure requires emission factors to become functions of farm management and conditions. Process-based models are suitable to explore specific functions. They are not suitable to identify the wide range of consequences that farm management options have on these emission processes which go beyond the biophysical processes they describe. This also indicates the complexity in identifying the trade-offs between GHG emissions. From the biophysical viewpoint they are the best option to explore such trade-offs, from a farm levels perspective they are too narrow and may not be suitable to identify the overall consequences and trade-offs that farm management options may impose, including for example the restriction imposed by manure legislation.

3.7. Intensity of ruminant production systems; stall-feeding versus grazing

Ruminant production systems may differ enormously in intensity of feeding and managing animals, varying from almost free ranging to full confinement systems feeding maize silage mostly. In deliverable D8.3 three show-case farms were studied which cover this range of system types. Despite the extreme differences in production, feeding and fertilisation intensity per hectare for these three farms, the on-farm GHG emissions did not differ dramatically when expressed per kg of milk produced. The contribution of individual GHG sources to the overall GHG budget was very different however. Furthermore, simulated size of emission sources differed strongly from IPCC Tier 2 estimates which makes clear that Tier 2 estimates cannot be used to evaluate and compare farming systems differing in level of management intensity on contribution of individual GHG sources to overall farm GHG budget and carbon footprint of

animal products. Enteric CH₄ estimates were 10 to 15% lower with the Dutch Tier 3 model than according to IPCC Tier 2 methodology on the farms with intensive feeding and a high proportion of maize products in the dairy diet (the farms with no grazing and restricted grazing). Furthermore, N₂O emissions predicted by the DNDC model were much higher than according to IPCC Tier 2 methodology for all farms, whereas C sequestration was higher on the farms with intensive feeding and lower on the farm with unrestricted grazing, particularly with the low N fertilisation regime.

The detailed effects simulated by process-based model simulations appear strongly related to the farming conditions and management. This warrants models to be used for farm surveys or for GHG inventories should accommodate GHG emission factors that are more condition-dependant. At least generally confirmed effects have to be represented if the impact of farm management and adaptation to climate change on the mitigation potential of farm measures has to be accounted for. The effect of N fertilisation on soil C sequestration or the effect of high dietary proportions of maize products or the effect of grassland management on enteric CH₄ are clear examples of aspects that should be accounted for instead of treating them compliant with higher Tier approaches. When comparing grassland-based systems with stall-feeding and confinement systems, many further aspects require detailed evaluation. For example, effects of housing on emissions from manure and floors, ability to control and optimize nutrition to minimize emission with stall-feeding, optimize utilisation of stored manure for optimal soil fertilisation, achieving higher feed intake and efficiency of feed utilisation with confined systems, or the impact of grazing animals on pastures and C stocks in soils are aspects that would require a detailed analysis of consequences for GHG emissions when comparing mechanisms of trade-offs between GHG emissions.

3.8. Increasing stocking density for increased grassland use efficiency

The final measure to be discussed is the impact of increased stocking density of animals to increase grassland use efficiency. Improved stocking density has a high impact on the nutritional value of the grass consumed. As discussed earlier, consumption of younger grass not only delivers more nutrients but also causes a lower enteric CH₄ emission per unit of grass consumed. However, this effect can easily be offset by a lower grass allowance because of the higher stocking density and hence lower intake level and productivity achieved. Exactly the latter situation was simulated for an Irish grazing study case where stocking densities of 4.5 to 6.5 lactating cows per hectare were compared (deliverable D8.3). For such high densities there were profound impacts on grass intake and milk yield per cow which overruled the CH₄ emission density per kg of milk produced leaving a limited effect of grass characteristics. However, the lower emission of enteric CH₄ per unit of milk may partly be offset again by a higher N₂O emission intensity if milk is produced and more N is excreted per hectare with the higher stocking density.

At far lower stocking densities (2.2 compared to 1.2) on the same Irish farm, combined with an adapted rate of artificial fertiliser N application according to grass requirement, was evaluated to have strongly reduced N losses, including lower N₂O emissions, due to the decrease in N fertilisation with stocking density from 226 to 156 kg N ha⁻¹ year⁻¹ (deliverable D8.3; Bannink *et al.*, 2014). A trade-off of the lower stocking density and lower N fertiliser application was a reduced C sequestration rate. Another GHG trade-off that may occur is a higher enteric CH₄ emission per unit of grass ingested or per unit of milk produced because of a change in grass

(digestive) characteristics with decrease in N fertilisation. The lower stocking density with the lower N fertilisation rate, and as a result lower N excretion rate per hectare, may accentuate such differences in grass characteristics. Based on model simulations it is expected that the lower the N fertilisation rate and the lower the stocking density (*i.e.* the more extensive the grazing system), the more likely GHG emission per kg of milk will be affected because of changed grass characteristics, and the higher the relative importance of the trade-off towards more enteric CH₄. Recent *in vivo* measurements of CH₄ emissions with varying grass herbage quality reported by Warner *et al.* (2015) confirm this importance (Figure 3.2).

It is questionable whether methodologies of GHG inventory or farm monitoring actually can accommodate for such effects and quantify the net results with the various trade-offs between CH₄, N₂O and C sequestration, and the milk yield achieved. In respect of the need to be able to identify important trade-offs between GHG sources, it is of high priority to improve the methodologies in this respect and to make the trade-offs visible. Also in view of the possible increase in grass production as a result of a changed climate (*i.e.* higher temperature and CO₂ concentrations) and a longer growing season, it is of interest to explore how to benefit from this by an increased stocking density, perhaps in combination with making grass stocks to overcome intermittent dry, warm seasons with insufficient grass growth that may incidentally occur. This could be an important adaptation option to climate change. To avoid an increase in GHG emissions by this adaptation process, stocking densities and farm management should be optimized. This seems only feasible if the models or methodology used to determine this optimal stocking densities and farm management are capable of tracking the effects on GHG emissions and trade-offs between them in sufficient detail.

4. The impact of climate change on simulated GHG emissions on three show-case farms varying in intensity of feeding, grazing, stocking density and N fertilisation

Livestock GHG emissions are dominated by CH₄ and N₂O emissions, with grasslands generally considered a CO₂ sink (Soussana *et al.*, 2007). The GHG balance of these systems is greatly affected by human activities (*e.g.* tillage, cultivation, irrigation and fertilization) and is a strong contributor to regional GHG budgets (Sogaard *et al.*, 2002). However, the GHG balance of these systems is also heavily influenced by climatic factors. There is an exponential relationship between ecosystem respiration and temperature (Lloyd & Taylor 1994), while N₂O and nitrate (NO₃) leaching are primarily driven by variation in soil moisture (Flechar *et al.* 2007). Also increases in wind speed and air temperature will result in increased ammonia volatilisation (Soegaard *et al.*, 2002). In order to assess the impacts of climate change on confinement (no grazed) and extensive grazed pasture systems, dynamic biogeochemical models were coupled to the outputs of global circulation models (GCM's).

4.1 Farm systems as case study

The farm systems investigated were the same farms used in Deliverable 8.3. Briefly, a Dutch high input, no grazing systems and a Dutch low input, restricted grazing system were (confinement) were compared with two Irish systems applying unrestricted pasture grazing (grazing) with a low or a high input (see Table 4.1 or Deliverable 8.3 for further details).

- 1) The Dutch high input system was a no grazing (total confinement), high fertilization, intensive dairy farming. This farm was in the SW Netherlands on a sandy soil with a high stocking density and high milk yield per hectare, without grazing (100% stall-feeding) and a high dietary proportion of maize silage and purchased concentrates. Only about a quarter of all maize silage fed is grown on-farm, the remainder being purchased.
- 2) The low input Dutch confinement system consisted of restricted grazing (mainly confinement), low fertilization/low emission, intensive dairy farming. This case involves the well-monitored experimental farm De Marke of Wageningen UR which is located at the most emission sensitive soil in the East of the Netherlands (dry sandy soil).
- 3) The high input Irish farm systems consisted of unrestricted grazing, extensive dairy farming with high fertilisation in Solohead, Co. Tipperary, Ireland (Humphreys *et al.*, 2009). Grazing management is principally practiced with only short periods of confinement during winter, relatively small amounts of concentrates purchased, and a relatively small proportion of animal excrements captured in manure storage. The intensive managed system had a high rate of N fertilizer applied and a grass sward mainly composed of perennial ryegrass.
- 4) The low input Irish farming system of unrestricted grazing was also in Solohead but with a less intensive management with a low rate of N fertilization and a grass sward with a relatively high proportion of clover.

Table 4.1. Characteristics of the four modelled farm systems.

Characteristics	Dutch, no grazing, high fertilization	Irish, grazing (high/ low fertilization)	Dutch, no (low) grazing , low fertilization
Area (ha)	38	11	55
Area grassland (% total area)	70	100	61
Fertilization rate (kg N/ha/yr)	447	96 / 226 (22 / 6% clover)	230
Dietary grass : maize silage	1 : 3.5	1 : 0	1.4 : 1
DM intake (kg DM/d)	19.9	14.6 / 14.3	18.5
Density (# lactating cows /ha)	3.4	2.2	1.5
Dietary concentrate (%DM)	32 (& 6 byprod)	10	21 (& 23 other)
Milk yield (kg/d per cow)	26.8	17.9	22.9

Source: project Cow & Opportunities (WUR Livestock Research) & Solohead, Teagasc (Humphreys et al., 2009)

4.2 Modelling and climate scenarios

Predicted values of future GHG emissions for the four farm systems were compared by driving the Dutch Tier 3 model (Bannink *et al.*, 2011) for CH₄ emissions and the DNDC model for N and soil C emissions using two sets of climate projections. These projections were generated using a Regional Climate Model (RCM) to dynamically down-scale the CGCM3.1 GCM from the Canadian Centre for Climate Modelling (EC., 2011) using RCP4.5 and RCP8.5 greenhouse gas emission scenarios. The initial low resolution climate forcing coming from earth system models (ESMs) was downscaled to a horizontal distribution of 0.5°. The data were also bias corrected using statistical bias observed during the historical period. Simulations were run over a reference period, 1990-2005 (baseline), and a future period, 2005-2099 (future scenario). The Representative Concentration Pathways, RCP4.5 and RCP8.5, are two of the four greenhouse gas concentration trajectories adopted by the IPCC for its Fifth Assessment Report (AR5) (Moss *et al.*, 2008) and represent an intermediate and the most intensive warming scenario, respectively. Climate projections used in this study were the latest available climate scenarios for North West Europe at the time of the study. They represent a projection of the current economic pathway and the temperature and precipitations are a result of the economic forcing. A CO₂ concentration of 370 ppmv was used in the DNDC model for baselines and an annual increasing of 2 ppmv was applied for the future scenarios (IPCC, 2007). The results of the future periods were compared to historical results generated from CGC3.1. Differences between the two periods gave an estimate of climate change.

4.3 Impacts of future climate change on reactive N emissions and soil organic carbon

In general, the grazed pasture systems exhibited greater vulnerability to climate change compared to the confinement systems. This was due to the fact that wetter spring and autumn conditions were predicted for Ireland (20% higher precipitation by 2099) and partly due to the fact that grazed systems are more coupled to the prevailing climate conditions. Increased N₂O and N leaching were observed for grazed systems under the RCP 4.5 scenario with a 16% and 21% increase in N₂O observed for the high N and low N systems respectively (Table 4.2). Under RCP 8.5, these emissions increased by 22% and 35% respectively. Similarly NO₃ leaching increased by 30% (high N) and 33% (low N) under RCP 4.5 and 38% (high N) and 41% (low N) under RCP 8.5 (Table 4.2). In contrast, N₂O and NO₃ emissions for the confinement systems only rose, on average, by 12% and 13% respectively. This was due to the fact that animals were kept indoors so decisions on manure and fertiliser application were far more controllable. By contrast, 70% of the increase in N emissions in grazed systems were due to increased pasture, range and paddock emissions. Carbon sequestration was also sensitive to high temperature and soil moisture deficits and was highly variable. Grasslands appeared more vulnerable, with up to 51% decrease observed in annual SOC sequestration in both Irish grassland systems. By comparison, Dutch managed systems exhibited a 21% reduction, possibly due to a larger proportion of maize/cereals in the cultivation mix (Table 4.2).

Ammonia emissions also exhibited an increase under RCP 4.5 AND 8.5 for the Dutch confinement systems (Table 4.2). This was principally due to the larger proportion of stored slurry in these systems. In contrast, led slurry was available in the grazed systems, so there was considerable less NH₃ emissions (Table 4.2).

In general, increases in emissions from confinement systems appear to be lower than for grazed pastures, although absolute level of emissions (per hectare basis) were higher for these systems. This occurred due to the relative lack of control in terms of N deposition in grazed systems particularly in terms of pasture paddock and range emissions (N₂O and leaching associated with urine and dung deposition).

We conclude that although the DNDC model successfully predicted SOC and GHG fluxes croplands and grasslands, key uncertainties can occur which were mainly due to poor characterisation of soil wilting point and field capacity. These characteristics govern the available water and hence nitrate leaching and N₂O emissions. Also, the crop growth module of the model is the application of a sigmoid curve based upon degree days and requires more parameters, such as base temperature as well as the degree days of phenology stages and radiation use efficiency to accurately define the growth curves for all crops in terms of their temporal carbon uptake. However, the annual summary of soil respiration and final crop yield were robust and hence the annualized SOC matched observations. Therefore, we can use the model for future annual predictions. The uncertainty in predicting the climate projections remains high however, and more work is needed on the climatic models used to generate future scenarios, and the underlying emission scenario assumptions.

In conclusion, future climate change is predicted to increase reactive N emissions by 24% in no-grazed and restricted grazing Dutch farming systems, and by 48% in Irish unrestricted grazing systems, with increase driven by wetter soil conditions in autumn and spring

Carbon sequestration exhibited high inter-annual variability under both RCP4.5 and 8.5 scenarios, driven principally by changes in summer temperature and precipitation. Larger decreases were observed for RCP 8.5 due to wetter winters and warmer summer.

Table 4.2. Mean GHG emissions and soil organic carbon sequestration for two Dutch confinement systems (with no grazing or restricted grazing) and two extensively managed Irish systems with unrestricted grazing. Emissions data has been generated using DNDC 9.4 and outputs from GCM's using RCP 4.5 and 8.5 climate scenarios.

Treatment	Statistics	Historical				RCP 4.5				RCP 8.5			
		1990-2005				2099				2099			
		NH ₃	N ₂ O	NO ₃	dSOC	NH ₃	N ₂ O	NO ₃	dSOC	NH ₃	N ₂ O	NO ₃	dSOC
		kg N ha-1 yr-1		kg C ha-2 yr-1		kg N ha-1 yr-1		kg C ha-2 yr-1		kg N ha-1 yr-1		kg C ha-2 yr-1	
No Grazing	Mean	63.1	5.75	97.8	211	70.3	6.47	102.2	184	79.3	6.59	96.5	185
(High Fertilisation)	Variance	11.3	1.69	17.8	150	17.4	1.98	32.4	485	31.8	2.47	39.7	211
Restricted Grazing	Mean	32.8	2.14	34.6	382	41.6	2.44	47.9	398	44.9	2.47	50.4	308
(Low Fertilisation)	Variance	7.8	1.14	15.9	224	17.6	1.76	24.1	393	13.8	1.85	30.1	207
Unrestricted Grazing	Mean	35.4	8.09	35.7	675	32.1	9.67	50.7	504	31.8	10.5	57.9	456
(High Fertilisation)	Variance	13.4	2.83	10.4	258	23.4	3.49	34.6	274	14.5	3.21	37.8	342
Unrestricted Grazing	Mean	17.8	4.74	17.4	498	16.8	6.06	25.9	402	17.2	7.34	29.8	329
(Low Fertilisation)	Variance	11.8	2.08	14.6	326	14.3	3.01	19	301	14.3	2.83	20.1	342

5. The effect of legumes on GHG emissions from European grasslands under climate change

5.1. Introduction

Grassland-based production systems result in three major GHG emissions – CO₂, N₂O and CH₄ – with fluxes closely linked with management practices, soil types and climatic conditions (Soussana *et al.*, 2004). Soil N₂O emissions result from microbial nitrate reduction (denitrification) and oxidation (nitrification) and are enhanced by N fertilization, atmospheric N deposition and biological N fixation by legumes (Mosier *et al.*, 1998). The magnitude of N₂O emissions also depends on environmental regulators (temperature, pH, soil moisture, that is, oxygen availability, and organic matter) which modify emissions at the time of N application (Dobbie *et al.*, 1999; Soussana, 2008). In grasslands, CH₄ emissions are dominated by enteric fermentation in ruminants and emissions from their effluents. Ruminant animals release approximately 5% of the ingested digestible C as CH₄ (e.g., Martin *et al.*, 2009). However, there is considerable variability in the magnitude of emissions due to both the animal characteristics (e.g. breed, age, production, physiological stage) and the diet (e.g. level of intake, feed processing, composition and interactions between components; Johnson and Johnson, 1995; Gworgwor *et al.*, 2006; Martin *et al.*, 2008; Seijan *et al.*, 2011).

A model-based assessment of GHG mitigation options at European grasslands was carried out using PaSim. Here, the objective was to provide emission/sink estimates of the major trace gases from grassland systems in Europe. Sustaining yields on the existing land base, whether under intensive pastoral systems production, or extensive grassland management, is critical to mitigating GHG emissions from agriculture. According to the IPCC (2007), the mitigation potential of agriculture could be as high as 5.5-6.0 Gt CO₂ eq. per year by 2030 of which approximately 1.5 Gt CO₂ eq. is from grazing land management (FAO, 2009). Therefore grasslands have a high potential to promote build-up of carbon (C) if appropriate management practices will be adopted. Plant litter and animal wastes supply grassland soils, which generally contain substantial amounts of organic carbon C. Grassland GHG fluxes from grasslands can, therefore, be partly mitigated by grassland C sequestration in soil organic matter (Soussana *et al.*, 2010).

Given the higher incidence of non-CO₂ emissions on the global warming effect, a reduction of N fertilisation can be considered as a mitigation option. Symbiotic fixation by legumes is an input to the N cycle. They have numerous features that can act together at different stages in the soil-plant-animal-atmosphere system, and these are most effective in mixed swards with a legume proportion of 30-50% (Lüscher *et al.*, 2014). Important opportunities for sustainable grassland-based animal production include: (i) increased forage yield, (ii) substitution of inorganic N-fertilizer inputs with symbiotic N₂ fixation, (iii) mitigation and facilitation of adaptation to climate change, as elevated atmospheric CO₂, warmer temperatures and drought-stress periods increase, and (iv) increased nutritive value of herbage and raise of the efficiency of conversion of herbage to animal protein.

A promising avenue to mitigation is the reduction in emissions that can be obtained by increasing clover content while consequently decreasing N fertilizer rates (D8.3). However, although reductions in GHG emissions from grasslands are seen as priorities, mitigation strategies should not reduce the economic viability of enterprises. In this respect, increasing legume proportion in grass-legume swards can be valuable as adaptation of European grasslands to climate changes because some evidence is provided (D4.1) that legumes species may perform better than grasses, in terms of biomass production, under drought conditions in Europe.

With the purpose of assessing the role played by increased legumes proportions to mitigate GHG emissions, a modelling exercise was performed at European scale, in which

agricultural management was manipulated (current management versus high legume proportion and low mineral N fertilization) and the impact on the three main GHG emissions was assessed under climate change conditions.

5.2. The impact model

PaSim (Ben Touhami *et al.*, 2013; Ma *et al.*, 2015) is a grassland-specific model to simulate water, C and N cycling in grassland systems at sub-daily time step. Microclimate, soil biology and physics, vegetation, herbivores and management are interacting modules. Simulations are limited to the plot scale. Animals are only considered at pasture (not during indoor periods). Photosynthetic-assimilated carbon is either allocated dynamically to one root and three shoot compartments (each of which consisting of four age classes) or lost through animal milking, enteric methane (CH₄) emissions and returns, and through ecosystem respiration. Accumulated aboveground biomass is either cut or grazed, or enters a litter pool. The N cycle considers N inputs to the soil via atmospheric deposition, fertilizer addition, symbiotic fixation by legumes, and animal faeces and urine. The inorganic soil N is available for root uptake and may be lost through leaching, volatilization and nitrification/denitrification, the latter processes leading to nitrous protoxide (N₂O) gas emissions to the atmosphere. Management includes organic and mineral N fertilizations, mowing, and grazing, with parameters set by the user or optimized by the model.

A parameterization of PaSim was used as developed by Ben Touhami (2014) for European grasslands in the Bayesian formalism, and initialized via a spin-up process reusing the in situ weather input. In particular, soil pools were initialized to steady-state by running the model over tenths of loops of available meteorology following Lardy *et al.* (2011). PaSim was run at each pixel to simulate daily values of net ecosystem CO₂ exchanges, NEE (NEE = GPP-RECO, where GPP is gross primary production, RECO is ecosystem respiration), as well as N₂O and CH₄ emissions. Outputs for the three GHGs were presented as yearly cumulated values. For NEE, negative values indicate the system is a source of C losses, while positive values indicate that the system sequesters C from the atmosphere.

5.3. Climate and management drivers

Thanks to the ISI-MIP project (<https://www.pik-potsdam.de/research/climate-impacts-and-vulnerabilities/research/rd2-cross-cutting-activities/isi-mip>), it was possible (D2.3 and D2.4) to access a set of climate projections for different climate models and four Representative Concentration Pathways (RCPs). The initial low resolution climate forcing coming from earth system models (ESMs) was downscaled to a horizontal distribution of 0.5°. The data were also bias corrected using statistical bias observed during the historical period. Two climate models and two scenarios were selected for the purposes of AnimalChange. For the scenarios, RCP4.5 and RCP8.5 represent an intermediate and the most intensive warming scenario, respectively. For the climate models, IPSL-CM5 and HADGEM-2 have contrasted responses to the same radiative forcing curve. Data are available for the period 1951 to 2099. The analysis was performed on two reference periods to represent past and future climate conditions: 1951-2004 (past), 2005-2099 (future). The management practices for European grasslands are based on regional/national statistics derived by the economic model CAPRI (Klumpp *et al.*, 2014). We have assessed changes in the yearly values of NEE (kg C m⁻²), and N₂O (kg N m⁻²) and CH₄ (kg C m⁻²) emissions under both current clover proportions on the swards and increased values as re-calculated by CAPRI (which implied corresponding reduction of N fertilizer inputs). Spatial patterns in Europe are rendered out in



the form of smoothed, pixelated European maps (25-km grid) produced with NASA's Panoply Data Viewer, from NASA-GISS (<http://www.dataone.org/software-tools/panoply-data-viewer>), in which pixels are not weighted for the land area covered by grasslands.

5.4. Sensitivity to climate exposure

Grassland GHG fluxes are highly sensitive to heat waves and severe droughts that may turn grasslands into C sources (Ciais *et al.* 2005; Soussana *et al.* 2007). To quantify the probability for grassland systems to incur potentially hazardous climate events, precipitation and temperature hazardous events in each year were quantified via the agro-climatic metric of aridity by De Martonne (1942) (≥ 0 , with aridity represented by near zero values). The sensitivity to aridity was assessed by estimating GHG outputs of grasslands for arid years in both reference periods. In particular, arid years were represented by values of the aridity index falling into the bottom 25% of all calculated aridity values (that is, below the 25th percentile), taken from both periods 2005-2009 and 1951-2004.

5.5. Results

5.5.1. Spatial patterns of aridity

The results of the mean difference between future and past climate, rendered out in the form of smoothed, pixelated European maps (Figure 5.1), indicate a general climate shift towards an increased exposure to heat and drought stress, with the only exception of some areas of central-eastern Europe and the British islands with RCP 4.5 represented by IPSL-CM5.

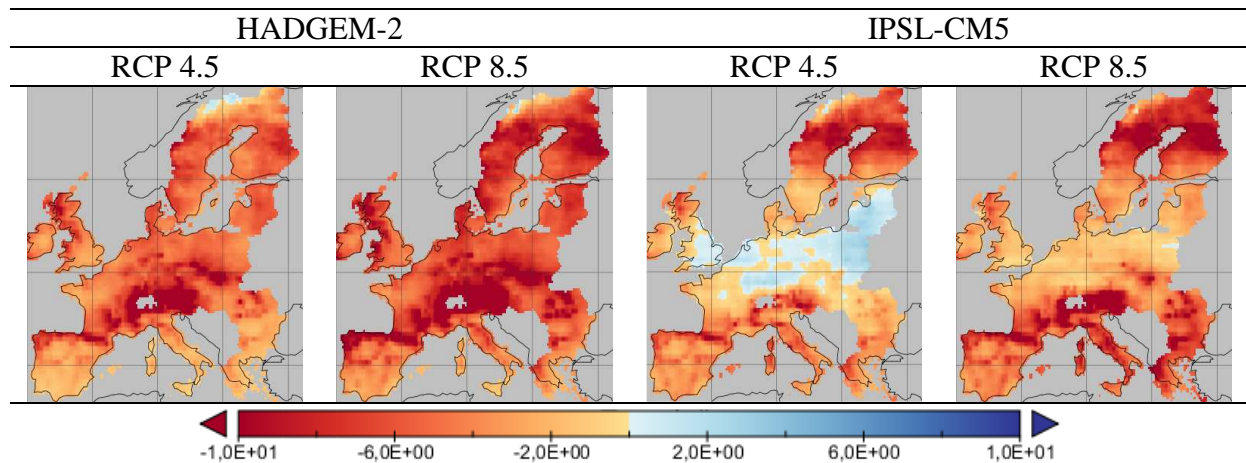


Figure 5.1. Difference between the mean values of the aridity index (b) calculated for the years 2005-2009 and 1951-2004 with $b < 25^{\text{th}}$ percentile, as represented by two climate models and two RCPs. Red colours indicate growing aridity under future climate (and vice versa for blue colours).

5.5.2. Mean GHG emissions

Mean continental values of the three GHG emissions under study are in Table 5.1, as simulated by PaSim for two reference periods (most arid years). Overall, the European grassland systems manifest a positive role in C sequestration (negative values of NEE). This is confirmed by projections, using alternative options of climate forcing and management. However, we estimate a reduction of the C sequestration potential in the future, more remarkable with HADGEM-2, RCP 8.5 and current management (~40%). This tendency is partly offset when legume-rich pastures are established.

For the N₂O and CH₄ emissions, this simulation study found that they are, in general, expected to increase in the future. For CH₄ emissions, an exception is represented by HADGEM-2 with RCP 8.5. For N₂O emissions, the estimated increase is about 40-80% over baseline climate and management (with the exception of HADGEM-2 and RCP 4.5, the presence of legumes in the sward can lessen this tendency).

Table 5.1. Means and variances of three output variables simulated by PaSim for two reference periods (most arid years) with two climate models, two RCPs and two management options.

Output	Management	Statistics	1951-2004	2005-2099			
				HADGEM-2		IPSL-CM5	
				RCP 4.5	RCP 8.5	RCP 4.5	RCP 8.5
NEE (kg C m ⁻² yr ⁻¹)	Current	Mean	0.0236	0.0066	-0.0029	0.0242	0.0255
		Variance	0.0006	0.0007	0.0006	0.0006	0.0005
	High legumes	Mean	0.0241	0.0074	-0.0020	0.0250	0.0263
		Variance	0.0006	0.0007	0.0006	0.0006	0.0005
N ₂ O (kg N m ⁻² yr ⁻¹)	Current	Mean	0.00064	0.00093	0.00116	0.00098	0.00118
		Variance	1.2E-07	1.9E-07	2.8E-07	2.2E-07	3.0E-07
	High legumes	Mean	0.00064	0.00095	0.00113	0.00095	0.00111
		Variance	1.2E-07	1.9E-07	2.8E-07	2.6E-07	2.2E-07
CH ₄ (kg C m ⁻² yr ⁻¹)	Current	Mean	0.001	0.0012	0.0004	0.0014	0.0014
		Variance	1.16E-07	1.72E-07	8E-06	2.12E-07	2.36E-07
	High legumes	Mean	0.001	0.0012	0.001	0.0012	0.0012
		Variance	1.16E-07	1.68E-07	1.68E-07	1.92E-07	1.88E-07

5.5.3. Spatial patterns of GHG emissions

Figure 5.2 presents different regional patterns of C source and sink distribution in European grasslands. HADGEM-2 driven projections depict NEE<1 values extending over central and western Europe up to the British Isles. IPSL-CM5 weather drivers have a lower impact, with scattered results. A weak impact of legume-rich management is apparent.

The results of Figure 5.3 indicate that there are hotspots of N₂O emission in European grasslands, mainly in Central-Northern Europe and more limited in the Mediterranean region. These hotspots tend to extend across larger areas according to future projections. Similar patterns are apparent among different climate and management scenarios (with limited impact of legume-rich management).

CH₄ emissions from European grasslands (Figure 5.4) also tend to cluster in hotspots, more numerous in the future with IPSL-CM5 scenarios (especially in central-northern Europe). Also in this case, changes in management do not have substantial effects on emissions.

It is noteworthy that, with high legume management, grasslands sequester C (Figure 5.2) and emit GHG fluxes (Figure 5.3, Figure 5.4) in areas of northern Scandinavia with climate change (with almost all scenarios), which is neither the case in the past nor in the projections with current management. This indicates that herbage is available and grazing may occur. The low temperatures experienced at such high latitudes are generally insufficient for plant growth. For the future, the resulting maps display that suitable conditions for high-legume swards are being generated with the new temperature conditions.

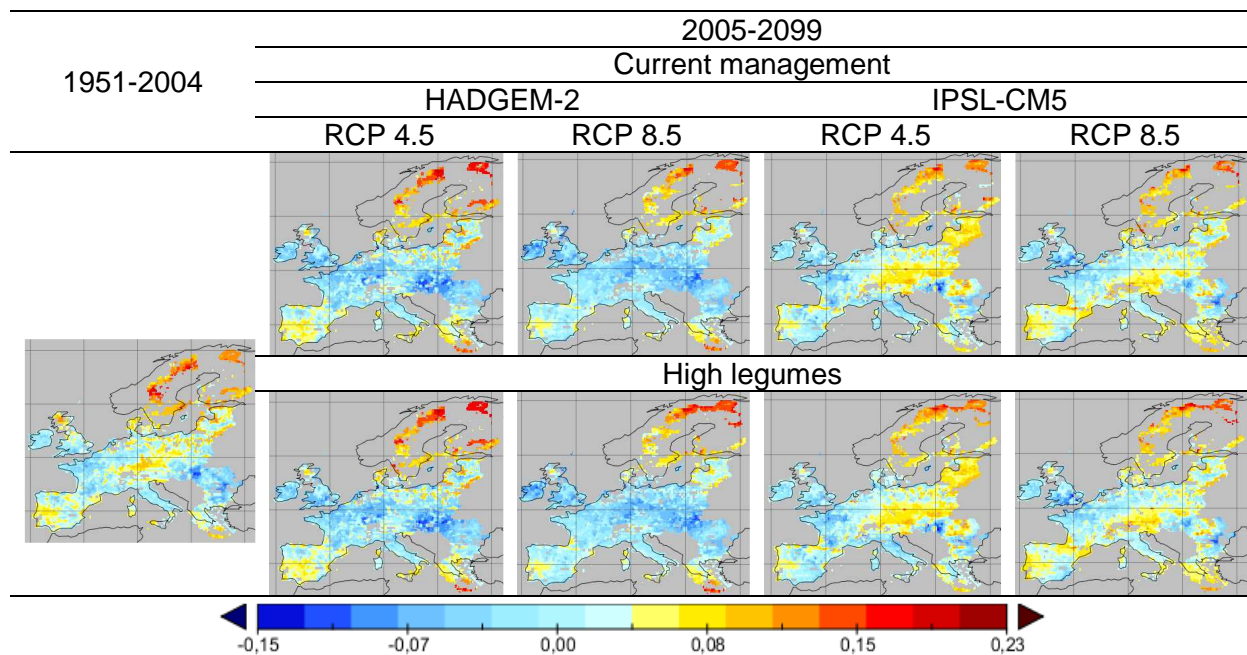


Figure 5.2. Average values of net ecosystem exchange ($\text{kg C m}^{-2} \text{yr}^{-1}$) for two time slices (most arid years), represented by two climate models, two RCPs and two management options. Light blue to dark blue colours indicate C emission (<0), light blue to dark brown colours indicate C sequestration (>0).

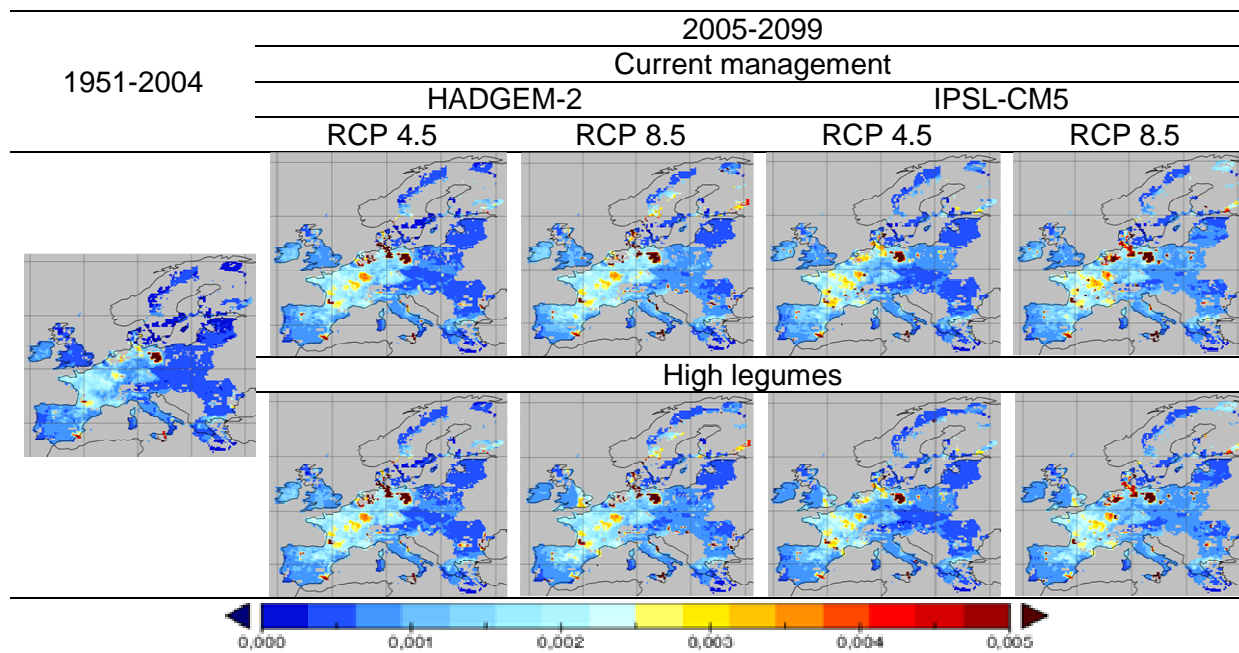


Figure 5.3. Average values of nitrous oxide emissions ($\text{kg N m}^2 \text{yr}^{-1}$) for two time slices (most arid years), represented by two climate models, two RCPs and two management options.

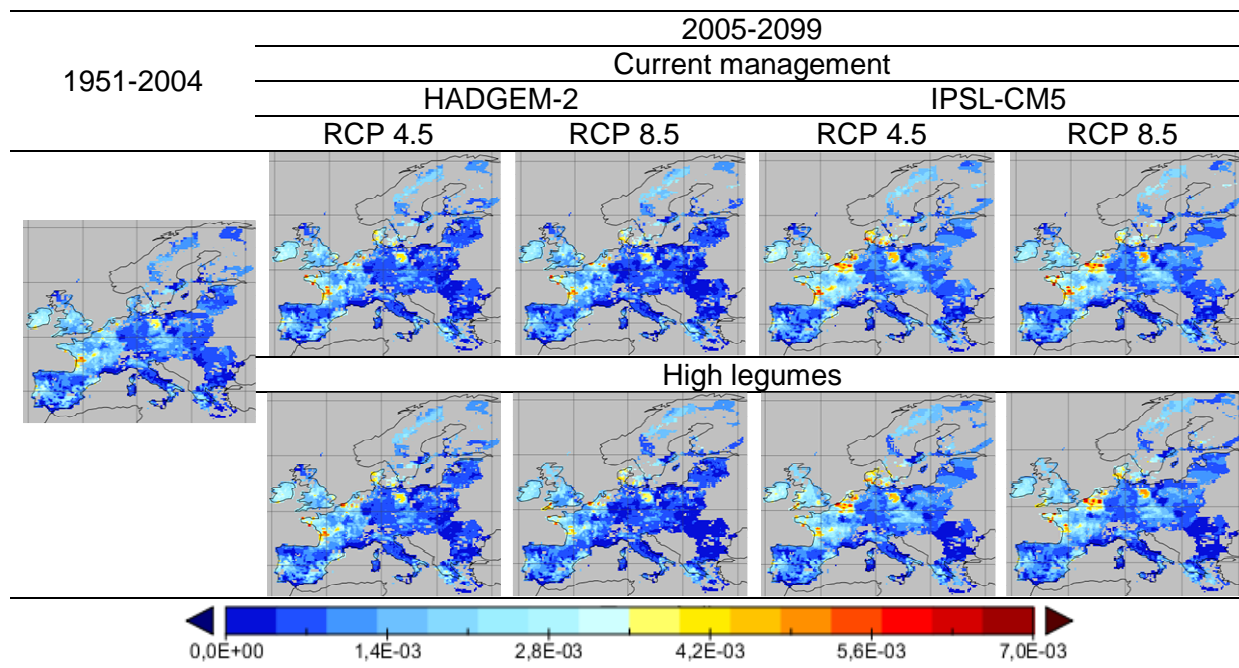


Figure 5.4. Average values of enteric methane emissions ($\text{kg C m}^2 \text{yr}^{-1}$) for two time slices (most arid years), represented by two climate models, two RCPs and two management options.

5.6. Conclusions on effects of legumes in European grasslands under CC

The modelling of GHG fluxes from grasslands is a non-trivial task, not only because of unsolved theoretical questions but also because fluxes are affected by large observational uncertainties. Being aware of these limitations, a large-scale assessment was performed of the GHG emissions from grassland systems to climate change. The study was focused on European grasslands, for which a parameterized version of the biogeochemical model PaSim is available along with detailed management inputs. The analysis was run to account for the sensitivity of grassland systems to arid conditions, which are expected to increase in the future in intensity, frequency and spatial extension, and can turn grasslands into C sources. The results of the simulations reliably show trends but absolute values are uncertain (e.g. N₂O emissions likely over-estimated) and require further studies. The study exhibits a highly contrasted spatial pattern of responses of GHG emissions. In particular, the results show that partly replacing N fertilization with increased proportions of legumes can slightly reduce CO₂, N₂O and enteric CH₄ emissions from European grasslands under future climate conditions, though this effect is not remarkable and differences are apparent among scenarios. This study also reflects the variability of management schemes in Europe. Interactions (mostly nonlinear) of climate features and N availability, due to different N rates applied to grasslands in different regions, can indeed explain most of regional differences (e.g. D5.3). Moreover, results refer to highly different grassland systems. Actually, country's boundaries may reflect the heterogeneity in the national/regional statistics since management data were built on national/regional statistics.

As a conclusion, an increase of legumes can be encouraged in the view of reducing GHG emissions under future climate conditions. Although a clear advantage of legume-rich swards in reducing GHG emissions from European grasslands has not been unambiguously disclosed by our results, some evidence of positive effects is provided. This evidence is a complement to the multiple benefits of legumes for the whole grassland-husbandry system, e.g. reduced dependency on fossil energy and industrial N-fertilizer, lower production costs, higher productivity, protein self-sufficiency (Lüscher *et al.*, 2014).



6. Discussion

This chapter discusses the general findings for the mechanisms underlying trade-offs between GHG emissions when adopting proposed mitigation measures on livestock farms under adaptation to climate change. Several important trade-offs mechanisms have been discussed that should be part of future studies on effects of climate change and efficacy of mitigation measures.

It was demonstrated that grassland management and N fertilization rates may have important consequences for grass characteristics, grass nutritive value and enteric CH₄ emission. Also a varying stocking density to optimize grass utilisation and N fertilization regime on soil GHG emissions may have important trade-offs with changes in soil N₂O emissions due to animal excreta and changed soil C sequestration. Furthermore, comparing (partial) confinement systems with unrestricted grazing requires the trade-off between higher GHG emissions from stored manure and stalled livestock with the confinement systems be weighed against the potential to exert more control on the utilisation of animal excreta in function of growing and weather conditions. Also the existence of feed stocks instead of relying on grazing solely may make farms less vulnerable to climate change or prevent unwanted degradation of grasslands under temporal conditions too harsh for grazing. The introduction of nitrification inhibitors (either soil applied, or animal applied) may have potential to circumvent negative trade-offs of management options that inflate N₂O emissions from animal excreta. Further model development is needed to be able to quantify the inhibition effect accurately and in function of details on farming conditions and management aspects (such as animal behaviour). An exchange between grass products and (forage) crop production may be an adaptation strategy to become less vulnerable for large fluctuations in feed stock or an increased seasonality of grazing under climate change; which may introduce the trade-off towards less or no C sequestration with crop production, risk of large C emissions due to crop rotations, but may have the synergistic effect of a introducing a less variability or uncertain feed quality and thereby more efficient animal production. Finally, the introduction of legumes is proposed as an adaptation strategy to make grassland and farming less vulnerable for the effects of extreme and warm weather conditions, and making grass production less dependent on N fertilizer inputs. A main trade-off would be however the higher vulnerability for excreta N emissions, lower nutritive value of legumes-rich grass swards or a lower yields and the need to supplement with other feeds, and less possibilities for farmers to anticipate on changes in grass characteristics which is detrimental for exerting control on feeding efficiency and animal productivity. An important synergistic effect would be that legumes reduce the requirement to purchase protein sources (and the GHG emissions associated with them) from external and less need for artificial fertilizer.

Studying the impact of climate change on GHG emissions and soil C revealed an increase in inter-annual variability of emissions, and an increase of reactive N emissions. The latter was particular prominent for the more extensive systems applying unrestricted grazing and fewer options for adapting farm management. Although there are positive effects of introducing legumes in grassland, making them less vulnerable to climate change in many areas of Europe, simulation results for higher proportions of legumes under more arid conditions did not clearly show positive effects of the introduction of legumes on simulated development of GHG. There seemed to be a marginal decline in emissions of CO₂, N₂O and CH₄ with increased legumes proportions. Heterogeneity in the response of European grasslands to climate change was very large, however, possibly reflecting their management.

All fore-mentioned trade-offs require detailed evaluation before they can be made conclusive and formulated in quantitative terms. Process-based models are in principal meant for such quantification task. It was shown before (Deliverable D8.3) that they be used



in conjunction, and can deliver more precise evaluation of GHG emissions in widely differing dairy farming systems. These models are able to simulate detailed response of GHG emissions to farm measures. Moreover these models have been evaluated against detailed empirical data at a totally different scale than that of the farming system. This makes these models are able to deliver mechanisms and insight in trade-offs, instead of relying on only empirical data for such trade-offs. Moreover, in methodologies for farm surveys or for GHG inventory basically stays unnoticed as a results of their construction. A combined use of process-based models and farm level methodologies is needed to ensure that progress can be made in identifying trade-offs between GHG emissions and include them in the search for mitigation options for livestock farms under various conditions and restrictions, with in some areas the need to adapt rapidly to the consequences of climate change.

It is highly relevant that such combined research activities are being acknowledged as strong trade-offs do occur when answers are to be found on questions related to changes in farming intensity (extensive vs. intensive) and changes in management of livestock farms under climate change (grazing, fertilization, stocking density, feed purchase, feed stocks, manure handling, circumventing the hazardous dry and hot seasons, managing soil C stocks, manure storage, controlling emission from animal excreta, managing feed supply, feed intake and animal productivity). This requires that not a choice is made between the type of model to develop or use, but that further model development should focus on both addressing options at the level of whole farm management, and the representation of underlying processes that generate the GHG emissions. If mechanisms of how farm management impacts on these underlying processes are covered well, farmers and policy makers also can find ways to find solutions and make these accountable and more transparent. An effective response of farmers to the effects of climate change and the need to mitigate GHG emissions warrants such transparency and methods which can quantify trade-offs detailed enough. As enteric CH₄ emission and soil N₂O emissions dominate farm GHG emissions, related process-based models should be able to contribute to further development of such methods.

7. Concluding remarks

This report provides a discussion of mechanisms involved in trade-offs between GHG emissions when potential mitigation measures on livestock farms. Many trade-offs may occur between GHG emissions of animal, manure or soil origin. The size and direction of these trade-offs (or sometimes synergies) are intricately related to farm management and several important trade-offs have been identified in this report that would require attention when addressing mitigation on (grass-based) livestock farms under climate change. The biochemical, biophysical, biogeochemical process-based models representing the mechanism involved with GHG emissions are in principle well suited, or can be made suitable rather easily, to delineate the effects of mitigation measures on GHG emissions, and to quantify the background and variability of potential trade-offs. They should best be applied when improving methodologies for farm surveys or GHG inventories on accountability of important sources of variation in GHG and in size of trade-offs in particular (in contrast to continuing the use of generic higher Tier approaches). The models are most suitable as research tools given the conditions and with inputs well defined, but they seem less suitable however to identify the consequences of farm management options, to evaluate alternative management scenarios from various viewpoints (e.g. legislation, economically, logistically) and to develop strategies to let farming systems adapt to climate change.



References

- Bannink, A., J. Kogut, J. Dijkstra, J. France, E. Kebreab, A.M. Van Vuuren, S. Tamminga, 2006. Estimation of the stoichiometry of volatile fatty acid production in the rumen of lactating cows. *Journal of Theoretical Biology* 238: 36–51.
- Bannink A., J. France, S. Lopez, W.J.J. Gerrits, E. Kebreab, S. Tamminga, J. Dijkstra, 2008. Modelling the implications of feeding strategy on rumen fermentation and functioning of the rumen wall. *Animal Feed Science and Technology* 143: 3-26.
- Bannink, A., M. C. J. Smits, E. Kebreab, J. A. N. Mills, J. L. Ellis, A. Klop, J. France, and J. Dijkstra. 2010. Simulating the effects of grassland management and grass ensiling on methane emission from lactating cows. *Journal of Agricultural Science* 148:55–72.
- Bannink, A., M.W. van Schijndel, J. Dijkstra, 2011. A model of enteric fermentation in dairy cows to estimate methane emission for the Dutch National Inventory Report using the IPCC Tier 3 approach. *Animal Feed Science and Technology* 166–167: 603– 618.
- Bannink, A., G. Bellocchi, A.-I. Graux, N. Hutchings, G. Lanigan, A. van den Pol-van Dasselaar, 2013a. First version of process based estimates of mitigation and adaptation options. Deliverable 8.2 of AnimalChange, 30 p.
- Bannink, A., J. L. Ellis, N. Mach, J.W. Spek, J. Dijkstra, 2013b. Interactions between enteric methane and nitrogen excretion in dairy cows. *Advances in Animal Biosciences* 4, s1: 19–27.
- Bannink, A., G. Lanigan, G. Bellocchi, N. Hutchings, A. van den Pol-van Dasselaar. 2014. Second version of estimates of mitigation and adaptation options determined with process-oriented modelling. Deliverable 8.3 of AnimalChange, 86 p.
- Ben Touhami, H., 2014. Calibration Bayésienne d'un modèle d'étude d'écosystème prairial : outils et applications à l'échelle de l'Europe. PhD Thesis, Blaise Pascal University, Clermont-Ferrand, France. (in French).
- Ben Touhami, H., Lardy, R., Barra, V., Bellocchi, G., 2013. Screening parameters in the Pasture Simulation model using the Morris method. *Ecological Modelling* 266, 42-57.
- Ciais, P., M. Reichstein, N. Viovy, A. Granier, J. Ogee, V. Allard, M. Aubinet, N. Buchmann, C. Bernhofer, A. Carrara, F. Chevallier, N. de Noblet, A.-D. Friend, P. Friedlingstein, T. Grünwald, B. Heinesch, P. Keronen, A. Knohl, G. Krinner, D. Loustau, G. Manca, G. Matteucci, F. Miglietta, J.-M. Ourcival, D. Papale, K. Pilegaard, S. Rambal, G. Seufert, J.-F. Soussana, M.-J. Sanz, E.-D. Schulze, T. Vesala, R. Valentini, 2005. Europe-wide reduction in primary productivity caused by the heat and drought in 2003. *Nature* 437: 529-533.
- De Martonne, E., 1942. Nouvelle carte mondiale de l'indice d'aridité. *Annales de Géographie* 51: 242-250 (in French).
- Dijkstra, J., H.D.St.C. Neal, D.E. Beaver, J. France, 1992. Simulation of nutrient digestion, absorption and outflow in the rumen: model description, *Journal of Nutrition* 122: 2239–2256.
- Dijkstra J., E. Kebreab, A. Bannink, L.A. Crompton, S. López, P.A. Abrahamse, P. Chilibroste, J.A.N. Mills, J. France, 2008. Comparison of energy evaluation systems and a

mechanistic model for milk production by dairy cattle offered fresh grass-based diets. *Animal Feed Science and Technology* 143: 203-219.

Dijkstra, J., J. France, J.L. Ellis, E. Kebreab, S. López, J.W. Reijs, A. Bannink. 2011. Effects of nutritional strategies on simulated nitrogen excretion and methane emission in dairy cattle. Pages 396-404. In: *Modelling Nutrient Utilisation and Digestion in Farm Animals*. Ed. D. Sauvant, J. van Milgen, P. Faverdin, N. Friggens. Wageningen Academic Publishers, Wageningen, The Netherlands.

Dijkstra, J., J. France, J.L. Ellis, A.B. Strathe, E. Kebreab, A. Bannink, 2013. Production Efficiency of Ruminants: Feed, Nitrogen and Methane., In: *Sustainable Animal Agriculture*, Ed. E. Kebreab, Pages 10-25. CAB International, Wallingford, United Kingdom.

Dobbie, K.E., I.P. McTaggart, K.A. Smith, 1999. Nitrous oxide emissions from intensive agricultural systems. Variation between crops and seasons, key driving variables and mean emission factors. *Journal of Geophysical Research* 104: 26891–26899.

Emberger, L., 1930. Sur une formule applicable en géographie botanique. *Comptes Rendus de l'Academie des Sciences de Paris* 19: 389-390 (in French).

FAO, 2009. *Grassland: enabling their potential to contribute to greenhouse gas mitigation*. Food and Agriculture Organization of the United Nations, Rome, Italy.

Flechard, C.R., A. Neftel, M. Jocher, C. Ammann, J. Leifeld, J. Fuhrer. 2007. Temporal changes in soil pore space CO₂ concentration and storage under permanent grassland. *Agricultural and Forest Meteorology* 142: 66–84.

Flechard, C.R., P. Ambus, U. Skiba, R.M. Rees, A. Hensen, A. van Amstel, A. van den Pol-van Dassel, J.F. Soussana, M. Jones, J. Clifton-Brown, A. Raschi, L. Horvath, A. Neftel, A.-I. Graux, 2011. *Modelling climate change impacts on grassland ecosystems. Ways to adapt forage systems*. PhD thesis, Blaise Pascal University, Clermont-Ferrand, France.

Graux, A.-I., G. Bellocchi, R. Lardy, J.-F. Soussana, 2013. Ensemble modelling of climate change risks and opportunities for managed grasslands in France. *Agricultural and Forest Meteorology* 170: 114-131.

Graux, A.-I., M. Gaurut, J. Agabriel, R. Baumont, R. Delagarde, L. Delaby, J.-F. Soussana, 2011. Development of the Pasture Simulation Model for assessing livestock production under climate change. *Agriculture, Ecosystems & Environment* 144: 69- 91.

Graux, A.-I., R. Lardy, G. Bellocchi, J.-F. Soussana, 2012. Global warming potential of French grassland-based dairy livestock systems under climate change. *Regional Environmental Change* 12: 751-763.

Gworgwor, Z.A., T.F. Mbahi, B. Yakubu, 2006. Environmental implications of methane production by ruminants: a review. *Journal of Sustainable Development in Agriculture & Environment* 2: 0794-8867.

Humphreys, J., I.A. Casey, A.S. Laidlaw, 2009. Comparison of milk production from clover-based and fertiliser-N-based grassland on a clay-loam soil under moist temperate climatic conditions. *Irish Journal of Agricultural and Food Research* 48: 189-207.



IPCC, 2006. 2006 IPCC guidelines for national greenhouse gas inventories. In: Greenhouse gas inventories programme, H.S. Eggleston, L. Buendia, K. Miwa, T. Ngara, K. Tanabe (Eds.), Institute for Global Environmental Strategies, Hayama, Japan.

IPCC, 2007. IPCC fourth assessment report: climate change 2007. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC. Cambridge University Press, Cambridge, United Kingdom and New York. New York, United States.

Humphreys, J., Casey, I.A., Laidlaw, A.S., 2009. Comparison of milk production from clover-based and fertilizer-N-based grassland on a clay-loam soil under moist temperate climatic conditions. *Irish Journal of Agricultural and Food Research* 48: 189-207.

Johnson, K.A., Johnson, D.E., 1995. Methane emissions from cattle. *Journal of Animal Science* 73: 2483-2492.

Klumpp, K., Martin, R., Eza, U., Leip, A., 2014. EU management file guide. UREP, INRA, Clermont-Ferrand, France and EC-Joint Research Centre, Institute for Environment and Sustainability, Monitoring Agricultural Resources Unit (MARS), Ispra, Italy.

Lardy, R., G. Bellocchi, J.-F. Soussana, 2011. A new method to determine soil organic carbon equilibrium. *Environmental Modelling & Software* 26: 1759-1763.

Le Houérou, H.N., 2004. An agro-bioclimatic classification of arid and semiarid lands in the isoclimatic Mediterranean Zones. *Arid Land Research and Management* 18: 301-346.

Li, H. J. Qiu, L. Wang, H. Tang, C. Li, E. van Ranst, 2010. Modelling impacts of alternative farming management practices on greenhouse gas emissions from a winter wheat-maize rotation system in China. *Agriculture, Ecosystems & Environment* 135: 24-33.

Li, D., J. Humphreys, G. Lanigan, 2011. Measured and simulated N₂O emissions from ryegrass and ryegrass/clover swards in a Moist Temperate Climate. *Plos-One* 6(10): e26176. doi 10.1371/journal.pone.0026176.

Lloyd, J., J.A. Taylor, 1994. On the temperature dependence of soil respiration. *Functional Ecology* 8: 315-323.

Lüscher, A., Mueller-Harvey, I., Soussana, J.F., Rees, R.M., Peyraud, J.L., 2014. Potential of legume-based grassland-livestock systems in Europe: a review. *Grass and Forage Science* 69, 206-228.

Ma, S., R. Lardy, A.-I. Graux, H. Ben Touhami, K. Klumpp, R. Martin, G. Bellocchi, 2014. On approaches and applications of Pasture Simulation model to simulate carbon and water exchanges in grassland systems. *Environmental Modelling & Software*, in press.

Ma, S., Lardy, R., Graux, A.-I., Ben Touhami, H., Klumpp, K., Martin, R., Bellocchi, G., 2015. Regional-scale analysis of carbon and water cycles on managed grassland systems. *Environmental Modelling & Software*, accepted.

Martin, C., D.P. Morgavi, M. Doreau, 2009. Methane mitigation in ruminants: from microbe to the farm scale. *Animal* 4: 351-365.



Martin, C., J. Rouel, J.P. Jouany, M. Doreau, Y. Chilliard, 2008. Methane output and diet digestibility in response to feeding dairy cows crude linseed, extruded linseed, or linseed oil. *Journal of Animal Science* 86: 2642–2650.

Mills J.A.N., J. Dijkstra, A. Bannink, S.B. Cammell, E. Kebreab, J. France, 2001. A mechanistic model of whole-tract digestion and methanogenesis in the lactating dairy cow: model development, evaluation and application. *Journal of Animal Science* 79: 1584–1597.

Mosier, A.R., J.M. Duxbury, J.R. Freney, O. Heinemeyer, K. Minami, 1998. Assessing and mitigating N₂O emissions from agricultural soils. *Climatic Change* 40: 7-38.

Moss R, M. Babiker, S. Brinkman, E. Calvo, T. Carter, J. Edmonds, 2008 Towards new scenarios for analysis of emissions, climate change, impacts, and response strategies (p. 132). Geneva: Intergovernmental Panel on Climate Change.

Reijs, J.W. 2007. Improving slurry by diet adjustments: A novelty to reduce N losses from grassland-based dairy farms. PhD Thesis. Wageningen University, Wageningen, the Netherlands.

Riedo, M., A. Grub, J. Fuhrer, 1998. A pasture simulation model for dry matter production, and fluxes of carbon, nitrogen, water and energy. *Ecological Modelling* 105: 141-183.

Schils, R.L.M, J.E. Olesen, A. del Prado, J.F. Soussana, 2007. A review of farm level modelling approaches for mitigating greenhouse gas emissions from ruminant livestock systems. *Livestock Science* 112: 240-251.

Schwinning, S., J. Parsons, 1996. A spatially explicit population model of stoloniferous N-fixing legumes in mixed pasture with grass. *Journal of Ecology* 84: 799–813.

Seijan, V., R. Lal, J. Lakritz, T. Ezeji, 2011. Measurement and prediction of enteric methane emission. *International Journal of Biometeorology* 55: 1-16.

Selbie D.R., K.C. Cameron, H.J. Di, J.L. Moir, G.J. Lanigan, K.G. Richards, 2014 The effect of urinary nitrogen loading rate and a nitrification inhibitor on nitrous oxide emissions from a temperate grassland soil. *Journal of Agricultural Science*, in press. (DOI: <http://dx.doi.org/10.1017/S0021859614000136>).

Søgaard, H.T., S.G. Sommer, N.J. Hutchings, J.F.M. Huijsmans, D.W. Bussink, F. Nicholson, 2002. Ammonia volatilization from field applied animal slurry: the ALFAM model. *Atmospheric Environment* 36: 3309–3319.

Soussana, J.-F., K. Pilegaard, P. Ambus, P. Berbigier, E. Ceschia, J. Clifton-Brown, S. Czobel, T. de Groot, J. Fuhrer, L. Horvath, A. Hensen, M. Jones, G. Kasper, C. Martin, C. Milford, Z. Nagy, A. Neftel, A. Raschi, R.M. Rees, U. Skiba, P. Stefani, S. Saletes, M.A. Sutton, Z. Tuba, T. Weidinger, 2004. Annual greenhouse gas balance of European grasslands—first results from the GreenGrass project. In: International conference greenhouse gas emissions from agriculture-mitigation options and strategies, 10–12 February, Leipzig, pp 25–30.

Soussana, J.-F., V. Allard, K. Pilegaard, P. Ambus, C. Ammann, C. Campbell, E. Ceschia, J. Clifton-Brown, S. Czobel, R. Domingues, C. Flechard, J. Fuhrer, A. Hensen, L. Horvath, M. Jones, G. Kasper, C. Martin, Z. Nagy, A. Neftel, A. Raschi, S. Baronti, R.M. Rees, U. Skiba, P. Stefani, G. Manca, M. Sutton, Z. Tuba, R. Valentini, 2007. Full accounting of the



greenhouse gas (CO₂, N₂O, CH₄) budget of nine European grassland sites. *Agriculture, Ecosystems & Environment* 121: 121-134.

Soussana, J.-F., 2008. The role of the carbon cycle for the greenhouse gas balance of grasslands and of livestock production systems. In: Rowlinson, P., Steele, M., Nefzaoui, A. (eds.) *Proceedings of the International Conference on Livestock and Global Climate Change of the British Society of Animal Science*, 17–20 May, Hammamet, pp 12–15.

Soussana, J.-F., K. Klumpp, T. Tallec, 2010. Mitigating livestock greenhouse gas balance through carbon sequestration in grasslands. *IOP Conference Series: Earth and Environmental Science* 6: 242-248.

Sutton, M.A., Tuba, Z., Weidinger, T., 2004. Annual greenhouse gas balance of European grasslands—first results from the GreenGrass project. In: *International conference greenhouse gas emissions from agriculture-mitigation options and strategies*, 10–12 February, Leipzig, pp. 25-30.

Thornley, J.H.M., 1998. *Grassland dynamics. An ecosystem simulation model*, CAB International, Wallingford, United Kingdom, 256 p.

Van den Pol-van Dasselaar, A., 2012. Qualitative overview of mitigation and adaptation options and their possible synergies & trade-offs. Deliverable 8.1 of AnimalChange, 32 p.

Vuichard, N., P. Ciais, N. Viovy, P. Calanca, J.-F. Soussana, 2007a. Estimating the greenhouse gas fluxes of European grasslands with a process-oriented model: 2. Simulations at the continental level. *Global Biogeochemical Cycles* 21: GB1005,1-13.

Vuichard, N., J.-F. Soussana, P. Ciais, N. Viovy, C. Ammann, P. Calanca, J. Clifton-Brown, J. Fuhrer, M. Jones, C. Martin, 2007b. Estimating the greenhouse gas fluxes of European grasslands with a process-oriented model: 1. Model evaluation from in situ measurements. *Global Biogeochemical Cycles* 21: GB1004,1-14.

Warner, D., S.C. Podesta, B. Hatew, G. Klop, H. van Laar, A. Bannink, J. Dijkstra. 2015. Effect of nitrogen fertilization rate and regrowth interval of grass herbage on methane emission of zero-grazing lactating dairy cows. *Journal of Dairy Science* 98. On-line first. <http://dx.doi.org/10.3168/jds.2014-9068>

