

# ANIMALCHANGE

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### THEME 2: FOOD, AGRICULTURE AND FISHERIES, AND BIOTECHNOLOGIES



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#### Abstract:

This report provides estimates of GHG emission factors for mitigation and adaptation options for intensive and extensive livestock farming. These were derived by process-oriented simulation models at animal and field level: the Dutch Tier 3 for enteric fermentation, PASIM and DNDC for soil and vegetation, and a newly developed Danish model for manure storage. Specific farm cases were evaluated by a combined use of the Dutch Tier 3 and the DNDC model. The impact of adaptation to climate change was studied using the PaSim model.

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Authors: André Bannink <sup>1</sup>, Gary Lanigan <sup>2</sup>, Gianni Bellocchi <sup>3</sup>, Nick Hutchings <sup>4</sup>, Agnes van den Pol-van Dasselaar <sup>1</sup>

<sup>1</sup> DLO, Wageningen UR Livestock Research, Lelystad, the Netherlands.

<sup>2</sup> Teagasc, Environment Research Centre Johnstown Castle, Environmental Research Department, Co Wexford, Ireland.

<sup>3</sup> INRA, Grassland Ecosystem Research Unit, Clermont-Ferrand, France.

<sup>4</sup> AU, Department of Agro-Ecology - Climate and Water Unit, Foulum, Denmark.

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Raphaël Martin, Haythem Ben Touhami & Katja Klumpp  
INRA, France.

Leon Šebek & Gerjan Hilhorst  
Dienst Landbouwkundig Onderzoek, Wageningen UR Livestock Research, the Netherlands.

James Humphreys & Pdraig O'Kiely  
Teagasc, Johnstown Castle & Grange, Ireland

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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 (CO<sub>2</sub>), and emissions of methane (CH<sub>4</sub>) and nitrous oxide (N<sub>2</sub>O) 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).

The present deliverable D8.3 provides estimates of mitigation potential by use of detailed process-oriented models and adaptation options and relates to Tasks 8.2 and 8.3 in WP8. In these tasks, process-oriented models were used, and if necessary adapted or improved, to evaluate the effect of mitigation measures under various conditions. A set of mitigation measures and conditions has been chosen which are relevant for studies on adaptation options to climate change. Task 8.2 is 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). In the present deliverable D8.3 further work on mitigation and adaptation options and evaluations of specific farm cases has been added.

For testing the effect of different options, mechanistic, dynamic models have been used: the Dutch Tier 3 for enteric CH<sub>4</sub> in dairy cows at the animal level, the PaSim and DNDC models for the field level (in PaSim: field and animal level), and a newly developed model at the manure storage level. Modelling results are described and discussed for promising options that have been identified in previous workshops 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). In the final year of AnimalChange, breakthrough mitigation options of WP6 and breakthrough adaptation options of WP7 will also be evaluated by the use of process-oriented models, where feasible. These results are not included in the present deliverable D8.3 however, but will be presented in deliverable D8.4.

For each model different options have been evaluated according to the characteristics of the model (animal level, manure pit level, field level). In the first version of the results of process-oriented models (D8.2; Bannink *et al.*, 2013a), emission estimates were given for the different components at the farm and field level. In the present deliverable D8.3 as the second version of process-oriented estimates of mitigation and adaptation options, the integral effect on these emissions is provided by combining the whole set of process-oriented models. Specific farm cases were selected with sufficiently detailed and reliable monitoring results. The farm cases were evaluated by a combined use of the Dutch Tier 3 and the DNDC model, and the newly developed model of manure storage. The impact of adaptation to climate change was thought to be studied best for the extensive case of livestock production. Therefore, PaSim model was used to simulate the effects of mitigation options on GHG emissions from extensive pasture-based ruminant production systems when adapting to various climatic scenarios.

Chapter 2 provides materials and methods for the use of the process-oriented models. Chapters 3 and 4 provide the effect of mitigation options on estimates of GHG emissions at the

field level and animal level, respectively. Due to differences in output of models used, in the aspects covered by the models and in the units used to express emission estimates, results are presented in different ways in the different chapters. Chapter 5 discusses the effect of adaptation of extensive pasture-based ruminant production systems on the potential to mitigate GHG emissions. Chapter 6 discusses the combined use of process-oriented models at the animal and field level for specific cases of intensive dairy farming, demonstrating the integral use of these models and the consequences for calculated on-farm net GHG budget. Finally, Chapter 7 discusses and integrates the results from preceding chapters, followed by concluding remarks in Chapter 8.

## 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 CH<sub>4</sub> 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 CH<sub>4</sub>. 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 CH<sub>4</sub> 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 CH<sub>4</sub>, the model calculates enteric CH<sub>4</sub> 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 CH<sub>4</sub> 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).

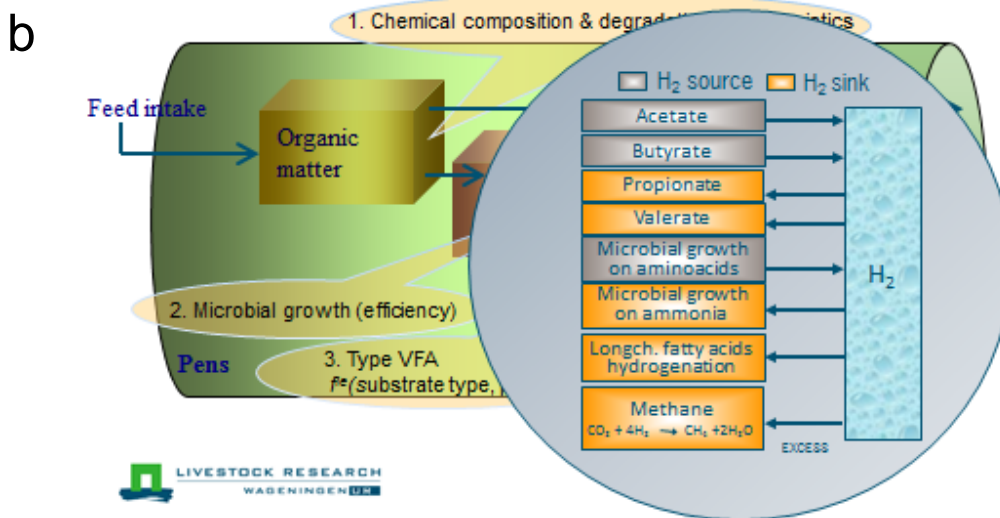
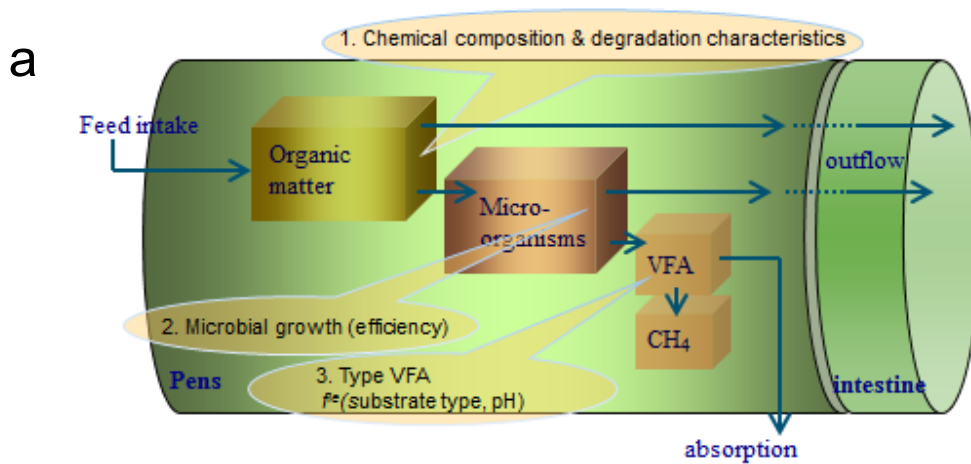


Figure 2.1. Diagram of the model structure of the Dutch Tier 3 for enteric fermentation in dairy cows, including three causal factors to explain variation in (a) microbial fermentation of feed substrate, microbial growth, production of volatile fatty acids and methane as end-products of fermentation, (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 CH<sub>4</sub>, CO<sub>2</sub>, ammonia (NH<sub>3</sub>), N<sub>2</sub>O, di-nitrogen (N<sub>2</sub>), hydrogen sulphide (H<sub>2</sub>S) 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.

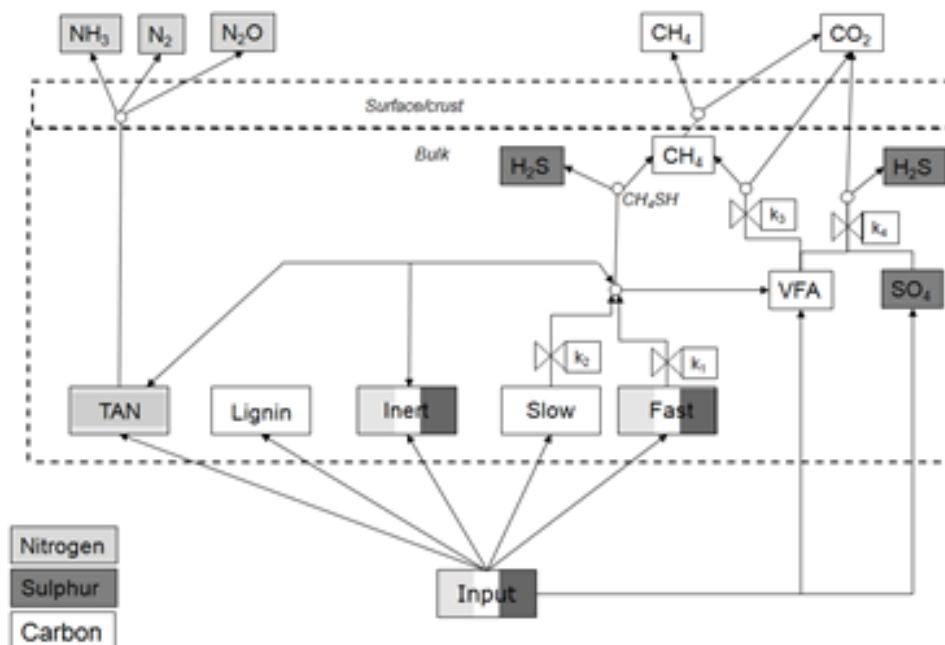


Figure 2.2. Diagram of a model of manure storage (Hutchings *et al.*, unpublished).

## 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](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 1/50<sup>th</sup> 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 m<sup>2</sup>) 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 CH<sub>4</sub> emission and returns, and through ecosystem respiration. Accumulated aboveground biomass is either cut or grazed, or enters a litter pool. Biological N<sub>2</sub> fixation is modelled according to Schwinning 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 N<sub>2</sub>O 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]).

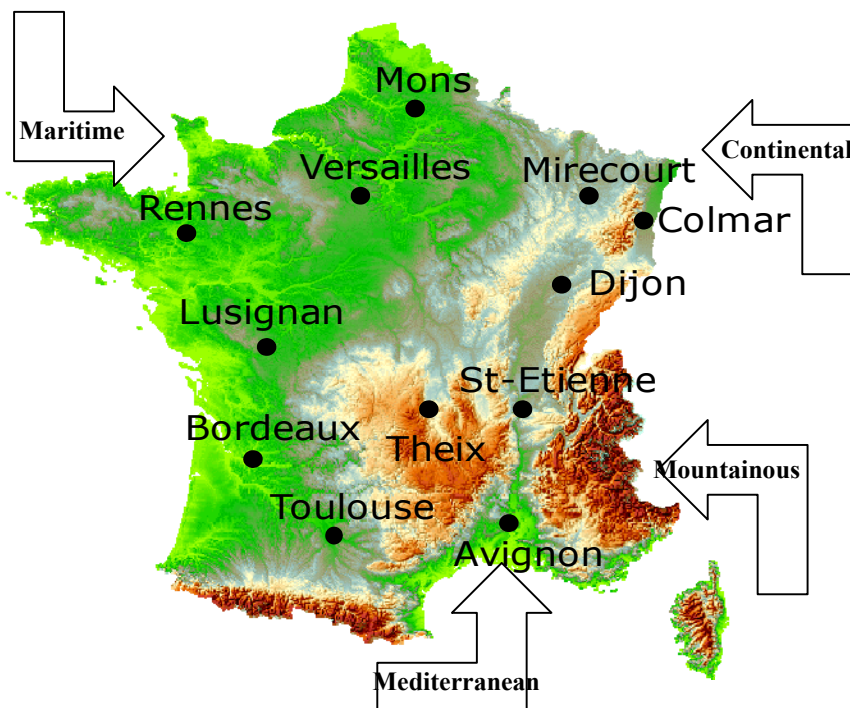


Figure 2.3. Location of 12 French study sites (from the ANR CLIMATOR project, [http://www.international.inra.fr/research/green\\_book\\_of\\_the\\_climator\\_project](http://www.international.inra.fr/research/green_book_of_the_climator_project)). Four sites were selected to represent contrasting agro-ecological 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	Raifall (mm yr <sup>-1</sup> )	T <sub>avg</sub> (°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 CO<sub>2</sub>, CH<sub>4</sub>, ammonia (NH<sub>3</sub>), nitric oxide (NO), N<sub>2</sub>O and dinitrogen (N<sub>2</sub>) 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 CO<sub>2</sub> 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 (NH<sub>4</sub><sup>+</sup>). The free NH<sub>4</sub><sup>+</sup> concentration is in equilibrium with both the clay-adsorbed NH<sub>4</sub><sup>+</sup> and the dissolved ammonia (NH<sub>3</sub>). Volatilization of NH<sub>3</sub> to the atmosphere is controlled by NH<sub>3</sub> concentration in the soil liquid phase and subject to soil environmental factors (*e.g.*, temperature, moisture, and pH). When a rainfall occurs, NO<sub>3</sub><sup>-</sup> 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, NH<sub>4</sub><sup>+</sup>, 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 N<sub>2</sub>O 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 (H<sub>2</sub>S) and CH<sub>4</sub> 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.

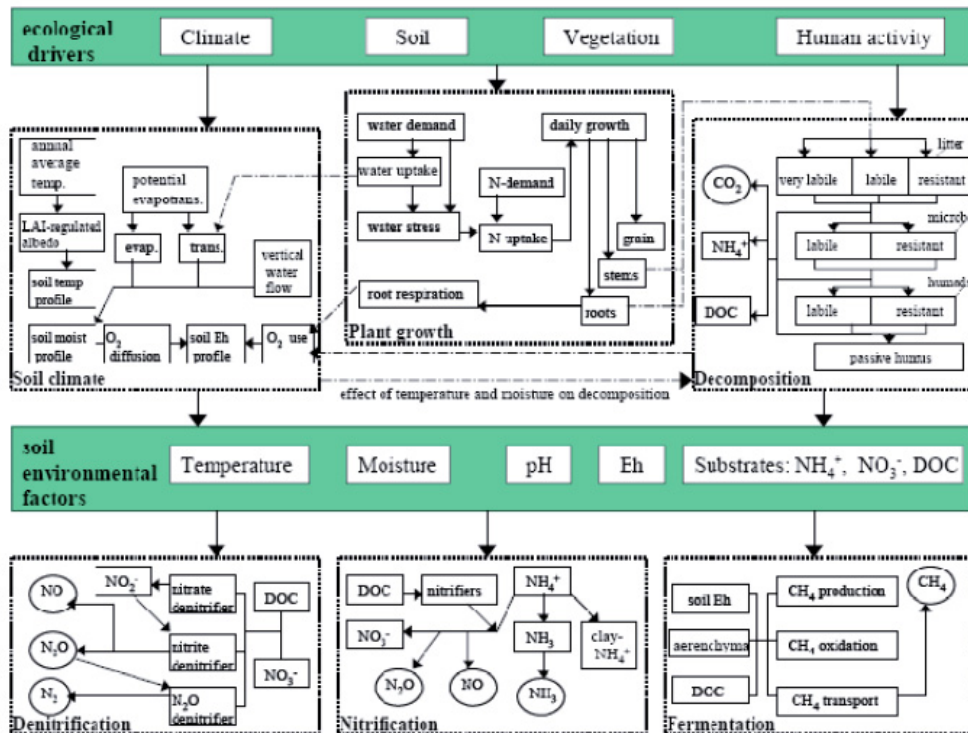


Figure 2.4. Diagram of model structure of DNDC, version 9.5 (Li et al., 2011).

### 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  $2.9 \text{ LSU ha}^{-1}$  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 ( $2.2 \text{ LSU ha}^{-1}$ ) 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 <sub>avg</sub>	Years
				(tC ha <sup>-1</sup> )	(mm yr <sup>-1</sup> )	(°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. Individual options at the field level

#### 3.1. Effect of nitrogen fertilisation rate on N<sub>2</sub>O emissions (PaSim & DNDC)

##### PaSim

The effect of N fertilisation rate on N<sub>2</sub>O emissions was tested with PaSim under contrasting agro-ecological zones in France. A monoculture perennial ryegrass (*Lolium perenne* L.) was simulated at four sites with 0, 100, 200, 300, 400 kg N ha<sup>-1</sup> for two cutting options (two [15/04, 15/08] and four [15/04, 15/06, 15/08, 15/10] cutting events per year).

The results of the effect of N fertilisation rate on N<sub>2</sub>O emissions are illustrated in Figure 3.1, in which:

- 1) Exponential increases of N<sub>2</sub>O emissions with N fertiliser rate were obtained, depending on initial SOM content and cutting frequency; this increase tends to become linear under arid conditions (e.g. at Avignon, top-right graph).
- 2) Similar increases of N<sub>2</sub>O emissions with N fertiliser rate generally were simulated under maritime, mountainous and continental climates; higher levels of N<sub>2</sub>O emissions occurred under Mediterranean conditions for humid years and continental conditions for arid years (in particular, Mediterranean conditions appear to be excessively emitting).
- 3) N<sub>2</sub>O emissions were lower for frequently cut grasslands established on organic-poor soils (intensive cutting tends to export more N from the plot, so that less N is available for denitrification and nitrification processes).

##### Avignon

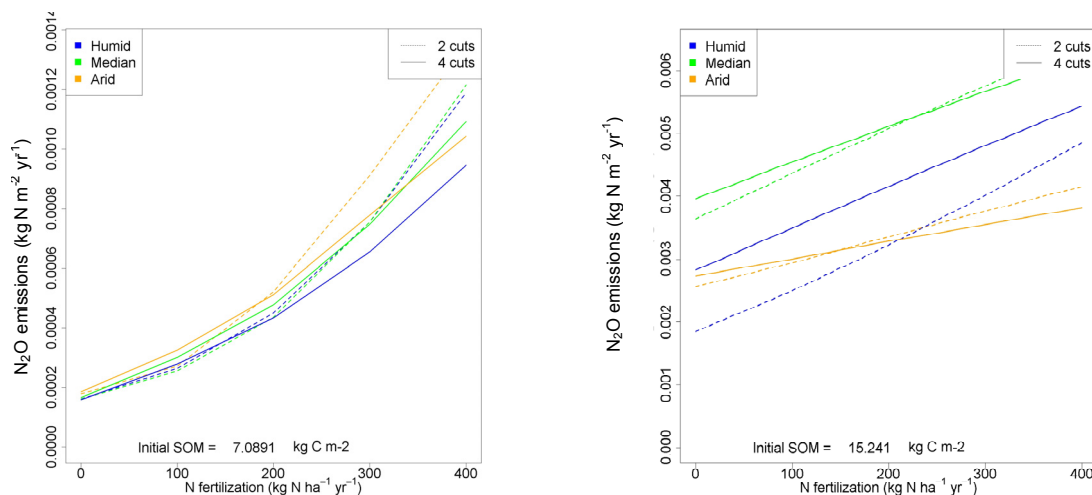
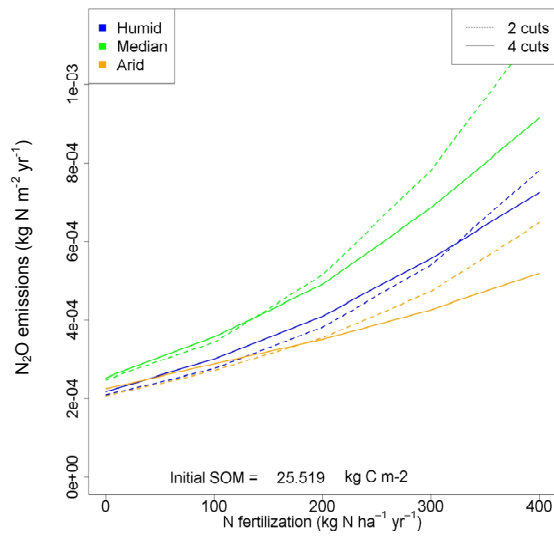
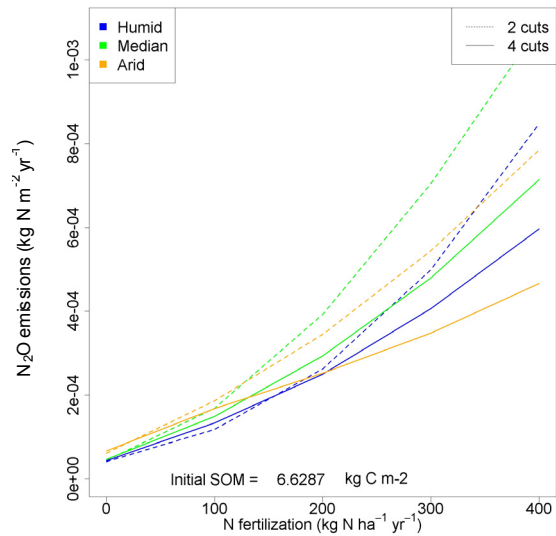


Figure 3.1. Annual N<sub>2</sub>O emissions simulated by PaSim at four French sites for contrasting years (from arid to humid) and for alternative soil organic matter (SOM) initializations, and cutting and nitrogen fertilisation regimes.

See next page for remainder of Figure 3.1

### Theix



### Mirecourt

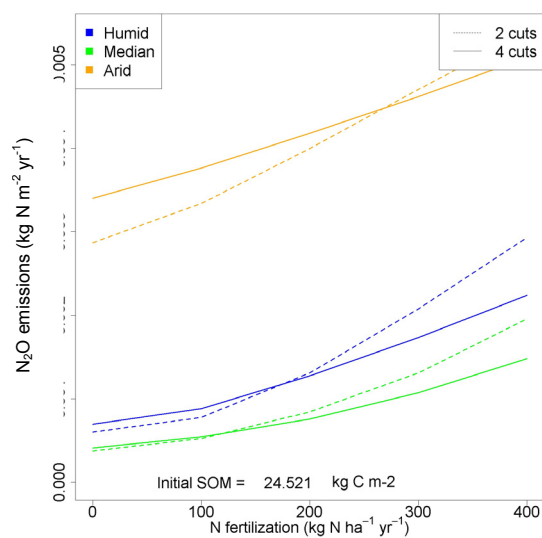
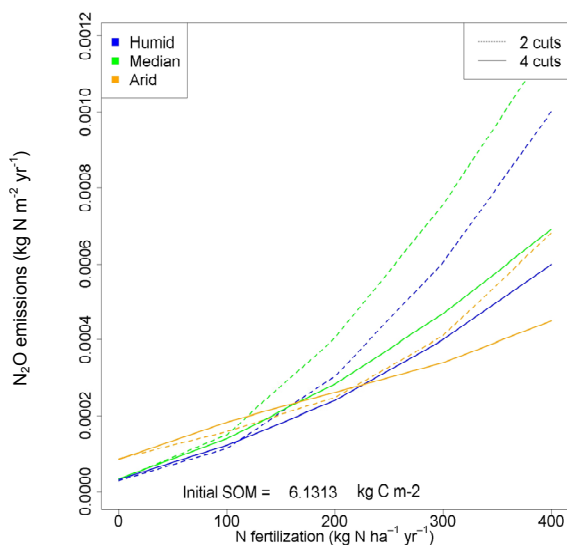


Figure 3.1. Annual  $N_2O$  emissions simulated by PaSim at four French sites for contrasting years (from arid to humid) and for alternative soil organic matter (SOM) initializations, and cutting and nitrogen fertilisation regimes.

See next page for remainder of Figure 3.1

## Rennes

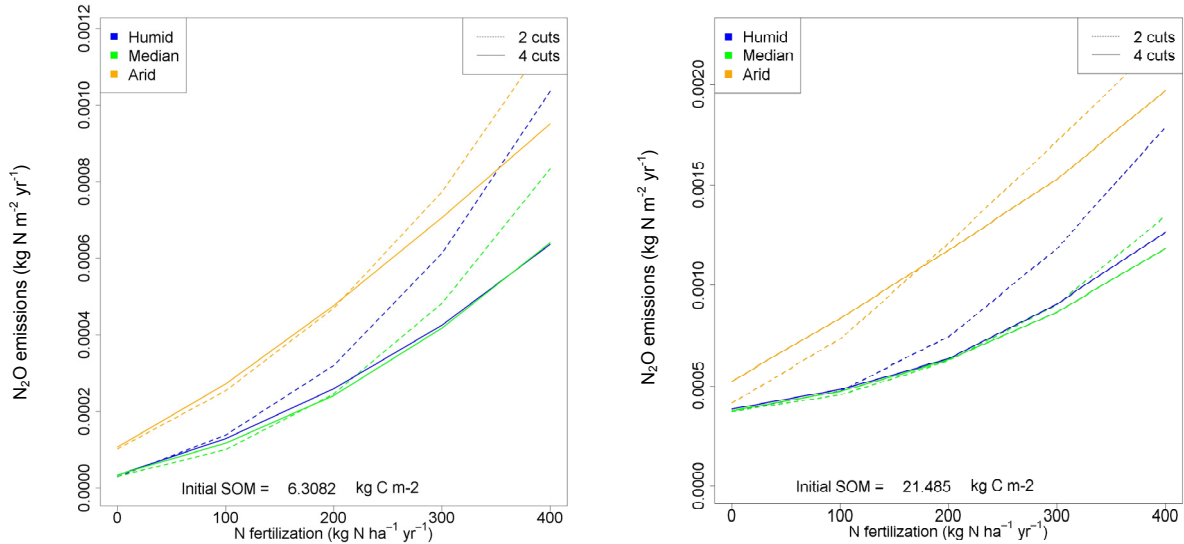


Figure 3.1. Annual N<sub>2</sub>O emissions simulated by PaSim at four French sites for contrasting years (from arid to humid) and for alternative soil organic matter (SOM) initializations, and cutting and nitrogen fertilisation regimes.

With respect to France, annual N fertilisation is usually about 100 kg N ha<sup>-1</sup> yr<sup>-1</sup> and does not exceed 200 kg N ha<sup>-1</sup> yr<sup>-1</sup>. In particular, Mediterranean grasslands are extensively managed. With the aim to mitigate N<sub>2</sub>O emissions, these simulations indicate that 1) N fertilisation on organic-rich soils needs to be limited, keeping it below 200 kg N ha<sup>-1</sup> yr<sup>-1</sup>, and 2) advantage needs to be taken of enhanced forage production due to temperature and CO<sub>2</sub> rises from climate change by increasing grass exports from the field (e.g. via cutting intensification).

With respect to the latter, a combination of warming, drought and elevated CO<sub>2</sub> may lead to important short-term N<sub>2</sub>O losses in extensively managed grasslands (Cantarel *et al.*, 2011). Questions still standing out are how to establish the maximum acceptable level for annual N<sub>2</sub>O emissions and what is the relationship between N<sub>2</sub>O emissions and N fertilisation rates under a variety of conditions. The reason for this is the difficulty to link N<sub>2</sub>O emissions with the aridity of climate as they closely depend on soil water content and soil temperature fluctuations (e.g. Flechard *et al.*, 2007).

## DNDC

Irish grasslands receive, on average, 100-250 kg N ha<sup>-1</sup> yr<sup>-1</sup>, with beef systems generally having lower N inputs compared to dairy systems (Schulte and Lanigan, 2010). As Irish livestock systems are almost exclusively grazed systems, pasture paddock and range emissions are included, this loading rate can reach up to 450 kg N ha<sup>-1</sup> yr<sup>-1</sup>. The effect of N application rate was modelled using DNDC 9.5 (Li *et al.*, 2000). N<sub>2</sub>O emissions were observed to increase exponentially for Johnstown (R<sup>2</sup> = 0.98) and linearly for Solohead (R<sup>2</sup>=0.97), while di-nitrogen emissions responded exponentially (R<sup>2</sup> = 0.96) to increasing N rate at both sites (Figure 3.2a). The change in the ratio of N<sub>2</sub>/N<sub>2</sub>O was surprisingly consistent across both sites, rising from >1 at 100 kg N ha<sup>-1</sup> application rate to 6.3 at 500 kg N ha<sup>-1</sup>. N<sub>2</sub>O was generally higher in the gleysols at Solohead compared to the loam-dominated Fluvisols at Johnstown. As a result of this soil type effect, leached N losses were 30% higher at Johnstown compared to Solohead at high N application rates. Both ammonia and leached N losses increased linearly between 200-500 kg N ha<sup>-1</sup> rates, with much lower emissions at the 100 kg N ha<sup>-1</sup> application rate due to the fact that almost all the N applied at this rate was taken up by the grass sward (Figure 3.2 c & d).

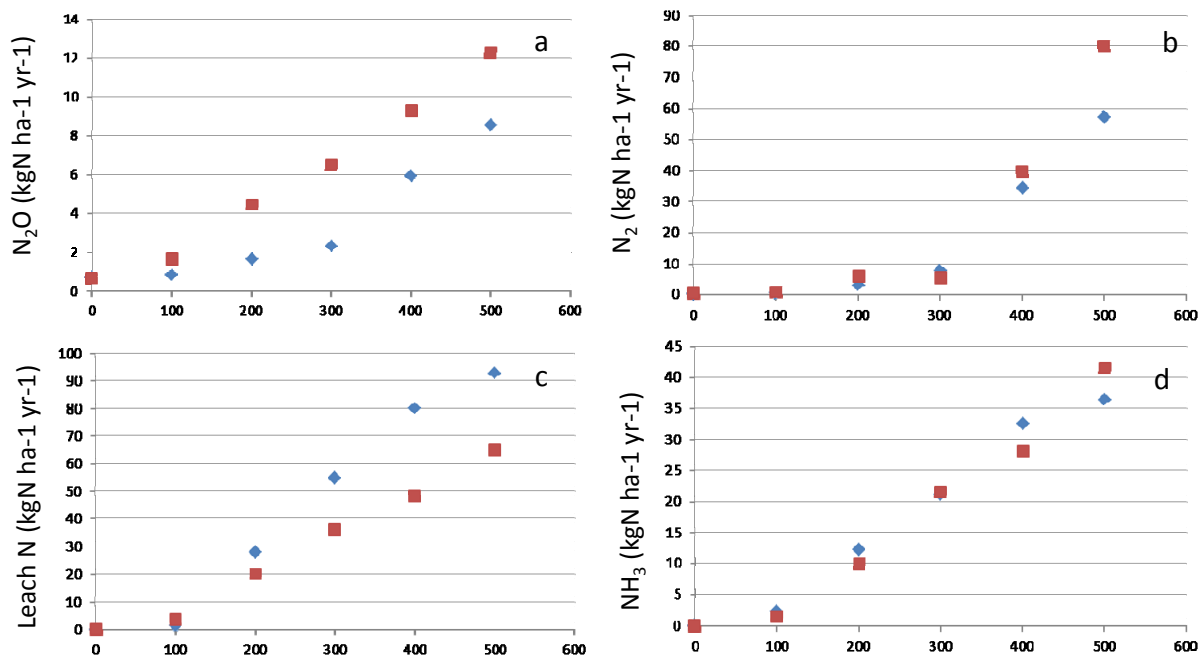


Figure 3.2. Annual a) N<sub>2</sub>O, b) N<sub>2</sub>, c) leached N and d) field NH<sub>3</sub> at Johnstown Castle (loam soil, 19% clay, 3.3% soil organic carbon; diamonds symbol) and Solohead (Gleysol, 27% clay, 4.5% soil organic carbon; squares symbol).

As a result of the differential N<sub>2</sub>O response to N application at both sites, the N<sub>2</sub>O emission factor was also observed to respond differently to N application rate and was different in absolute terms for Johnstown and Solohead (Figure 3.3a). In Solohead, there was a linear increase in emission factor up to 200 kg N application rate. At higher rates, the emission factor remained relatively constant at 2-2.35% of applied N. In comparison, emission factors were lower for Johnstown and increased linearly with application rate (R<sup>2</sup> = 0.93) and ranged from

0.1% at 100 kg N to 1.6% at 500 kg N. There was also a positive linear response of net biome productivity to increasing N application rate with net sequestration ranging between 316 kg C ha<sup>-1</sup> yr<sup>-1</sup> and 426 kg C ha<sup>-1</sup> yr<sup>-1</sup> for Solohead and Johnstown respectively (Figure 3.3b).

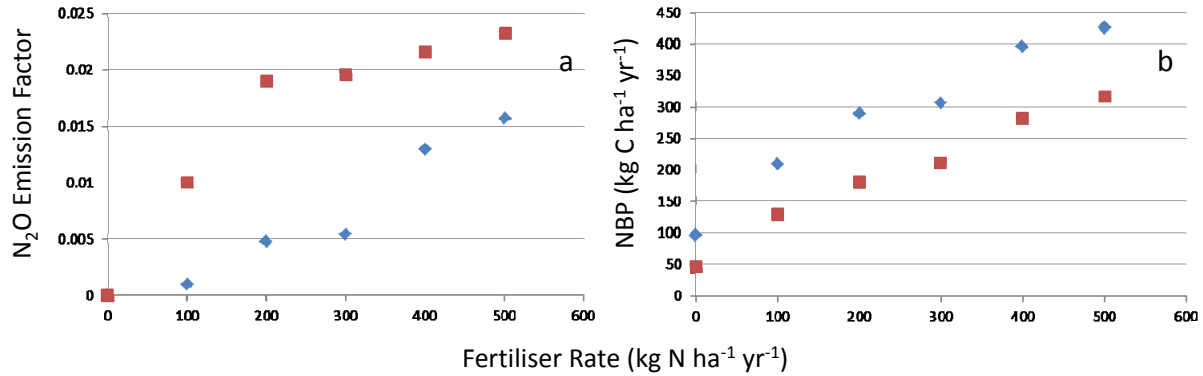


Figure 3.3. Annual a) N<sub>2</sub>O emission factor and b) net carbon biome productivity (NBP) at Johnstown Castle (loam soil, 19% clay, 3.3% soil organic carbon; diamonds symbol) and Solohead (Gleysol, 27% clay, 4.5% soil organic carbon; squares symbol).

### 3.2. Effect of legume fraction on N<sub>2</sub>O emissions (PaSim & DNDC)

#### PaSim

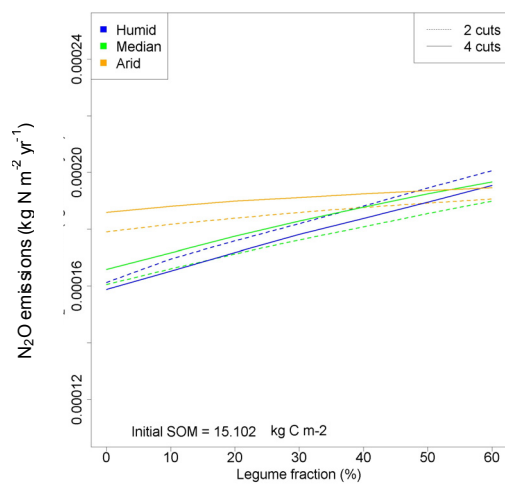
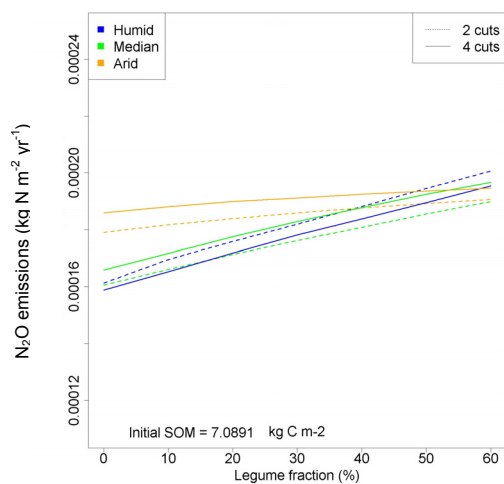
The effect of legume fraction on N<sub>2</sub>O emissions was tested with PaSim under contrasting agro-ecological zones in France. An unfertilised mixed sward of perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.) was simulated at two contrasting sites (Avignon, Mediterranean; Mirecourt, continental), either cut two (15/04, 15/08) or four times per year (15/04, 15/06, 15/08, 15/10), and containing either 0 or 10%, 20%, 30%, 40%, 50% and 60% of legume. The results are illustrated in Figure 3.4, in which:

- 1) An increase of N<sub>2</sub>O emissions was obtained in response to legume fraction, the progression changing from linear to exponential as moving from arid to humid climate conditions.
- 2) Simulated N<sub>2</sub>O emissions were higher under arid conditions.

Symbiotic fixation by legumes is an input to the N cycle. The magnitude of soil N<sub>2</sub>O emissions may thus depend on biological N fixation by legumes (e.g. Mosier *et al.*, 1998). The simulations indicate that N<sub>2</sub>O emissions are expected to rise with proportion of clover in grassland becoming higher than 20-30% (when focusing on humid and intermediate arid years). This proportion is therefore identified as an upper threshold for N<sub>2</sub>O mitigation purposes.

A limitation of the present study is that in the current version of PaSim the legume fraction is kept as a constant proportion in the sward, without a response to changing environmental and management conditions (e.g. cutting frequency, grazing pressure, water and nutrient availability, CO<sub>2</sub> concentration increase). Model improvements are in progress to clarify the dynamics of the legume component of a grass-legume mixture and incorporate this in the model representation.

## Avignon



## Mirecourt

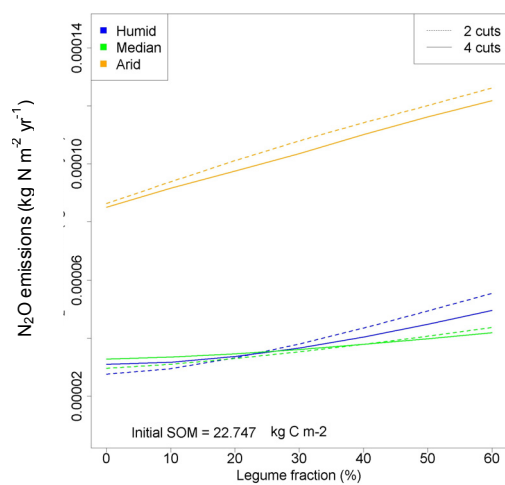
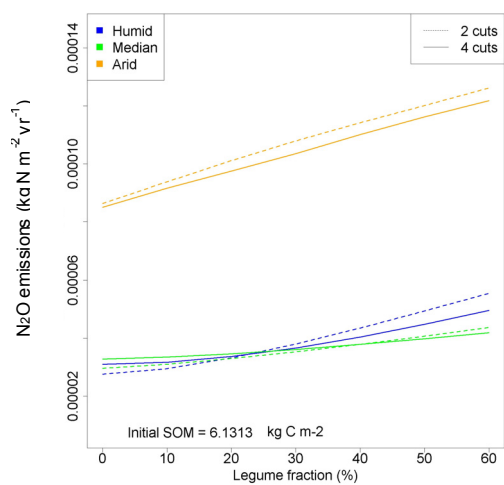


Figure 3.4. Annual N<sub>2</sub>O emissions estimated by PaSim at two French sites for contrasting years (from arid to humid) and alternative options of soil organic matter (SOM) initialization, cutting and nitrogen fertilisation.

## DNDC

With DNDC a conventional fertilised *Lolium perenne* monoculture and a grass/clover system was simulated for Johnstown and Solohead (see Table 1 for management details).

Table 3.1. Grazing and fertilisation (mineral fertiliser and slurry N application) for Solohead

	GG+FN	GWC+FN	GWC-FN	G-B	WC-B
Grazing	+ <sup>a</sup>	+	+	-	-
Slurry (m <sup>3</sup> ha <sup>-1</sup> ) and fertilizer N (kg N ha <sup>-1</sup> ) application in 2009					
Slurry (9 Feb) <sup>b</sup>	34 <sup>c</sup>	34	34	-	-
Urea (25 Mar)	28.4	28.4	-	-	-
CAN (8 May)	51	34	-	-	-
CAN (23 Jun)	26	-	-	-	-
CAN (14 Jul)	26	-	-	-	-
CAN (4 Aug)	26	-	-	-	-
CAN (1 Sep)	26	-	-	-	-
Slurry (m <sup>3</sup> ha <sup>-1</sup> ) and fertilizer N (kg N ha <sup>-1</sup> ) application in 2010					
Slurry (10 Feb)	28	28	28	-	-
Urea (14 Apr)	57.5	57.5	-	-	-
CAN (30 Jun)	33.8	-	-	-	-
CAN (22 Jul)	67.5	-	-	-	-
CAN (16 Aug)	67.5	-	-	-	-

The experiment was a randomized block design with five treatments and three replicates (Table 1). The treatments were: 1) grazed perennial ryegrass (*Lolium perenne*) swards receiving high N fertilisation (FN) rate (GG+FN), 2) grazed ryegrass/white clover (*Trifolium repens*) swards receiving low rate of N fertilisation (GWC+FN), 3) grazed ryegrass/white clover swards not receiving N fertiliser (GWC-FN), 4) perennial ryegrass plots (G-B) and 5) perennial ryegrass/white clover plots (WC-B). The swards (paddocks) of treatments GG+FN, GWC+FN and GWC-FN were rotationally grazed by dairy cows and have under the same treatment since the beginning of 2003 (GG+FN and GWC+FN) or 2008 (GWC-FN).

For the three grazing treatments, DNDC simulated N<sub>2</sub>O fluxes quite well in comparison with the measured fluxes during the non-grazing period for GG+FN (R<sup>2</sup>=0.86, P<0.001, n=20), GWC+FN (R<sup>2</sup>=0.82, P<0.001, n= 20) and GWC-FN (R<sup>2</sup>=0.81, P=0.05, n=20) (Figure 3.5). Although there were some discrepancies, significant correlation were found between the simulated and measured daily fluxes for GG+FN (R<sup>2</sup>=0.57, P<0.001, n=42), GWC+FN (R<sup>2</sup>=0.51, P<0.05, n=42) and GWC-FN (R<sup>2</sup>=0.42, P<0.05, n=42).

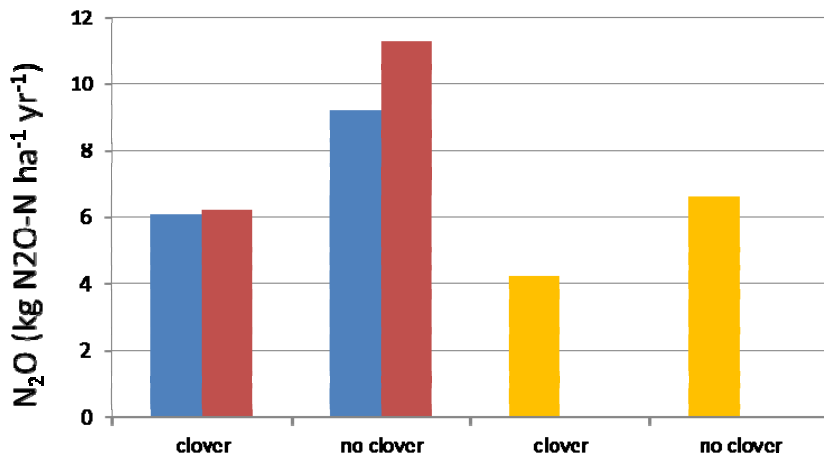


Figure 3.5. Modelled (blue; left) and measured (red; left) N<sub>2</sub>O emissions for Solohead and modelled emissions for Johnstown (orange; right).

The observed reduction in emissions is principally attributable to reduction in mineral fertiliser application. However, when clover addition is simulated with a uniform fertilisation rate, there is a discrepancy between measured and modelled emissions. Measurements indicate that for legume proportions between 20-40%, there is a drop in N<sub>2</sub>O emission factor, due to the optimisation of N usage between grass and legume (Figure 3.6). In contrast, modelled emissions continually rise with legume proportion and this is consistent for both PaSim (Figure 3.4) and DNDC (Figure 3.6). The discrepancy is due to the fact that a) DNDC treats biologically fixed N in a similar way to applied N and b) the model is unable to simulate N flows between grass and legume.

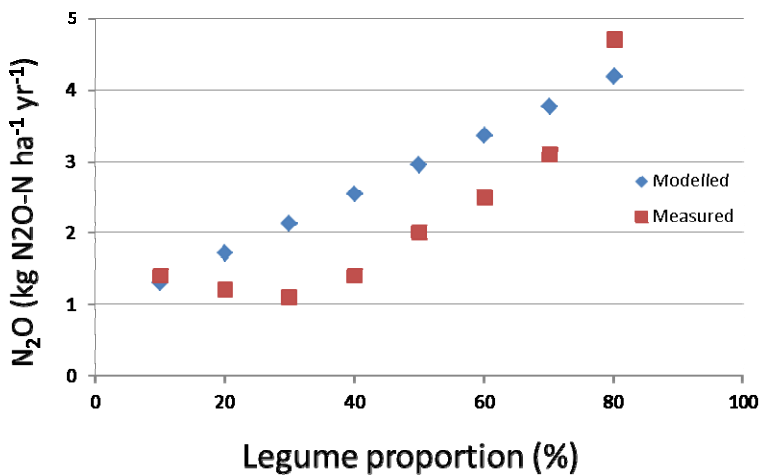


Figure 3.6. Measured and modelled N<sub>2</sub>O emissions for grass/clover mixtures with a varying proportion of clover in the grass sward.



### 3.3. Effect of stocking density / grazing period length on GHG emissions (PaSim)

The effect of stocking density and grazing period length on GHG (CO<sub>2</sub> equivalents from CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub>) emissions was tested with PaSim under contrasting agro-ecological zones in France. The sites of Theix (mountainous) and Rennes (maritime) were selected to represent two main production districts. The first corresponds to an upland area (Massif Central) of permanent pastures with suckling cattle. The second (located in Brittany, north-western France) matches farming systems with sown grasslands and dairy herds. For the latter, field grazing conditions were represented by dairy cows grazing a sown mixture of *Lolium perenne* L. and *Trifolium repens* L. To calculate GHG emissions (kg CO<sub>2</sub>-C eq. per unit area and per production unit), two management options were simulated: constant (without adaptation) and flexible (with adaptation). For the latter, an automatic procedure was activated to optimize stocking rate and grazing fractional coverage (Graux, 2011). Estimated “attributed net GHG” values, Att-NGHG, were evaluated, with Att-NGHG as an equation of the additive contribution of field and barn emissions for each GHG (CO<sub>2</sub>, N<sub>2</sub>O, CH<sub>4</sub>). PaSim only simulates on-site GHG-emissions. Off-site (barn) emissions were assessed according to IPCC guidelines (IPCC, 2006) and attributed to the corresponding grassland field under the assumption that harvested herbage is fully eaten by stalled cattle (Graux *et al.*, 2012). This third option was tested in combination with measures to adapt to climate variability.

Simulation results are presented in the form of exceedence probability distributions, calculated over a 30-year period from 1970 to 1999, for both the grazing length (Figure 3.7, upper graphs) and stocking density (Figure 3.7, lower graphs), and attributed net GHG per unit area (Figure 3.8; upper graphs) and per unit product (Figure 3.8; lower graphs).

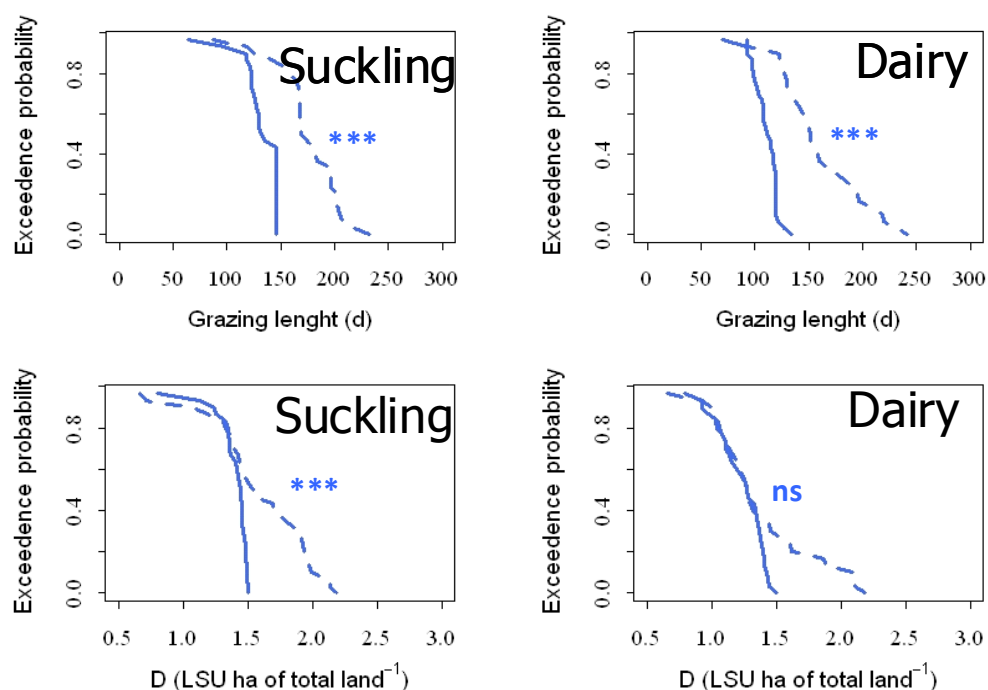


Figure 3.7. Exceedence probability distributions of grazing period length (d, top) and cow stocking density (D, bottom) (LSU, Livestock unit) for a **suckling cow system** (Theix, mountainous zone; left panel) and a **dairy farm** (Rennes, maritime zone; right panel) beef enterprises. Continuous line: constant management; dashed line: flexible management. \*\*\*:  $p < 0.001$ , ns:  $p \geq 0.05$  (Kolmogorov-Smirnov test).

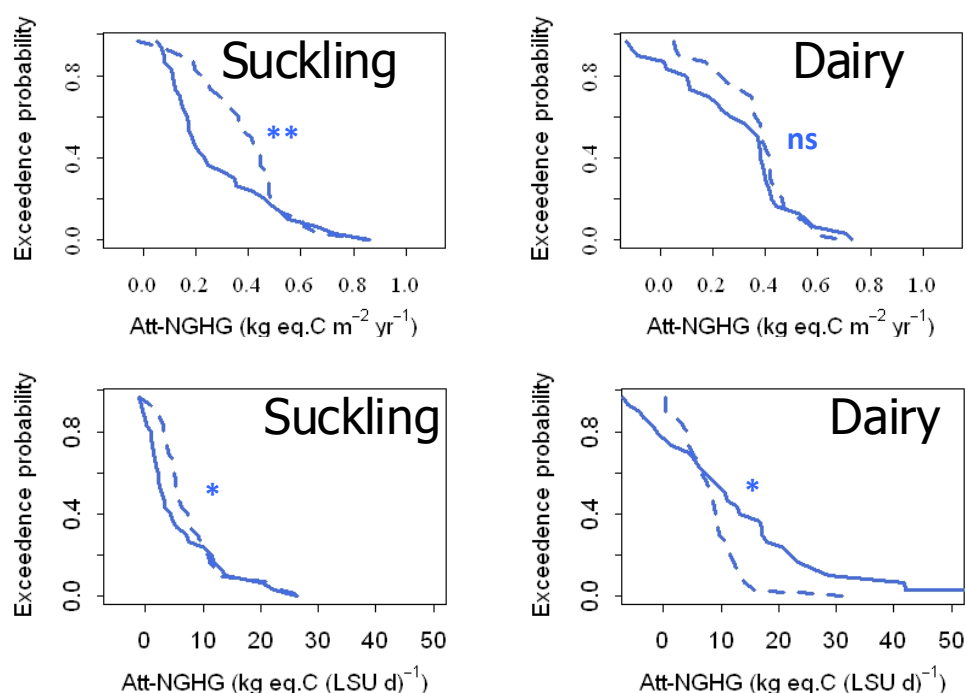


Figure 3.8. Exceedence probability distributions of attributed net GHG budget (Att-NGHG) for alternative grazing period lengths and cow stocking density for a **suckling cow system** (Theix, mountainous zone; left panel) and a **dairy farm** (Rennes, maritime zone; right panel). Att-NGHG is either given per unit area of the field (top) or per production unit (bottom). Continuous line: constant management; dashed line: flexible management. \*\*:  $0.001 < p < 0.01$ , \*:  $0.01 < p < 0.05$ , ns:  $p \geq 0.05$  (Kolmogorov-Smirnov test).

These preliminary results indicate that 1) an improved (flexible) management is needed (longer grazing time, higher stocking rate) to optimize grazing options with respect to year-to-year variability, and 2) with optimization, some additional risk of GHG emissions tends to be associated with suckling cattle systems in mountainous zones.

It would be interesting to run the model under projected conditions of climate change because both suckler and dairy livestock systems may benefit from the increase in annual herbage production as a result of a changed climate which is to be expected with higher temperature and CO<sub>2</sub> concentration. Increased herbage production would also allow an extended grazing period and increased stocking density. An adapted farm management may help to mitigate GHG emissions (either when expressed per unit area or per unit of product) while benefiting from increased availability of herbage.

### 3.4. Modelling reduction in stocking rate and slurry addition (DNDC)

Reduced stocking rate from 2.2 Livestock units (LSU) ha<sup>-1</sup> to 1.2 LSU ha<sup>-1</sup> lead to a reduction in total N<sub>2</sub>O emissions from 7.3 kg N<sub>2</sub>O-N ha<sup>-1</sup> yr<sup>-1</sup> to 4.9 kg N<sub>2</sub>O-N ha<sup>-1</sup> yr<sup>-1</sup> for Solohead and from 4.1 kg N<sub>2</sub>O-N ha<sup>-1</sup> yr<sup>-1</sup> to 2.9 kg N<sub>2</sub>O-N ha<sup>-1</sup> yr<sup>-1</sup> at Johnstown Castle due to a decrease in pasture paddock and range emissions and reduced fertiliser input from 226 kg N ha<sup>-1</sup> to 156 kg

N ha<sup>-1</sup> (Figure 3.9, Table 3.2). There was also a 12 kg NH<sub>3</sub>-N ha<sup>-1</sup> yr<sup>-1</sup> and 10 kg NH<sub>3</sub>-N ha<sup>-1</sup> yr<sup>-1</sup> reduction in volatilised N at Solohead and Johnstown Castle respectively as a result of reduced deposition and a 15–24 kg N reduction in leached N losses. However net CO<sub>2</sub> biome productivity (NBP) was also reduced by 0.08 t C ha<sup>-1</sup> yr<sup>-1</sup> and 0.1 t C ha<sup>-1</sup> yr<sup>-1</sup> due principally to a reduction in GPP. This, in turn, was due to reduced N input and alterations in defoliation patterns.

The addition of an extra 60 t slurry (or 90 kg TAN) was also simulated with DNDC for the high stocking rate treatment (note that mineral fertiliser is reduced by at the same time) with total N loading of 220 kg N ha<sup>-1</sup> yr<sup>-1</sup>. Direct N<sub>2</sub>O emissions and leached N losses were reduced due to the fact that most of the N was in organic or ammoniacal N form and was released more slowly than fertilisation with ammonium nitrate fertiliser and uptake of N in the crop was optimised. However, there was a 20% increase in volatilised N loss due to this increase in ammonium N loading. The addition of slurry also resulted in increased SOC levels of 0.2 t C ha<sup>-1</sup> yr<sup>-1</sup>. However, it should be noted that when this system was simulated over a 100 year period, SOC equilibrium was reached after 50 years. Therefore, the C sequestration if weighted for a 100 year period would be reduced to 0.1 t C ha<sup>-1</sup> yr<sup>-1</sup>.

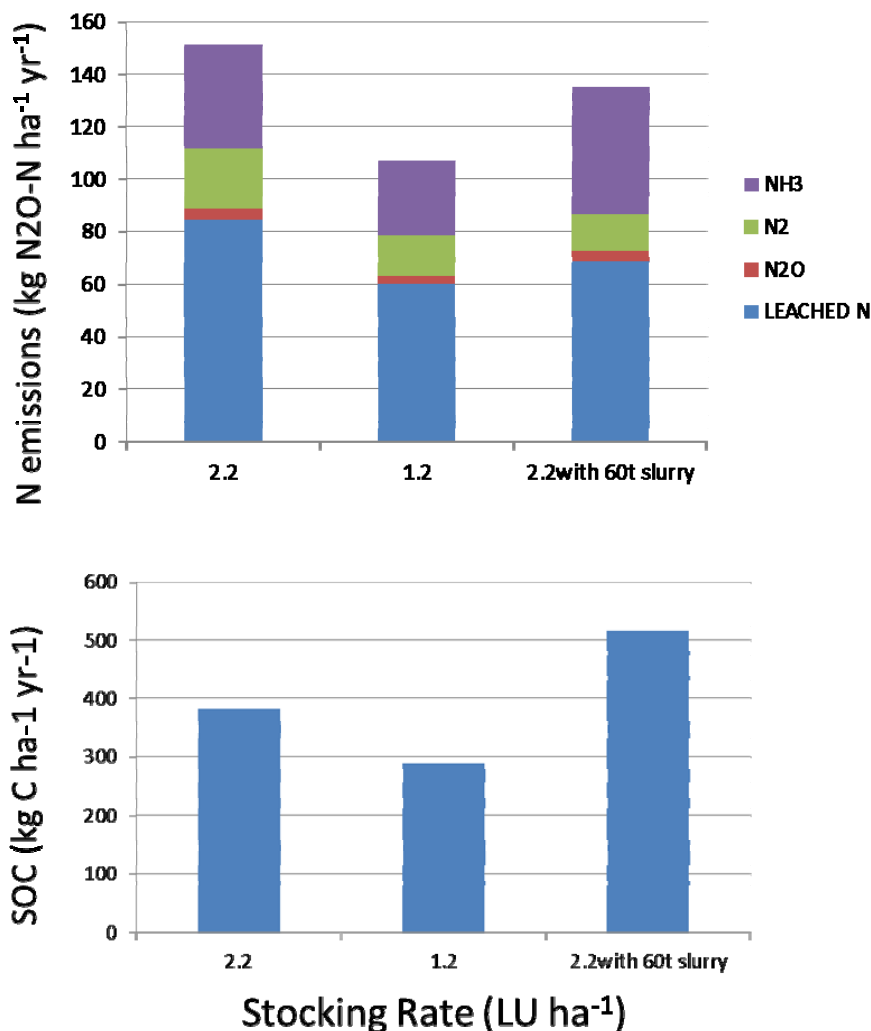


Figure 3.9. Nitrogen losses and soil organic carbon sequestration at two stocking rates (LU, livestock units ha<sup>-1</sup>) and with the addition of 60 t fresh weight of slurry.

### 3.5. Nitrification Inhibitors (DNDC)

The impact of nitrification inhibitors was also simulated (Figure 3.10). Inhibitor application occurred in March and September as these are considered the periods when reactive N losses will be highest. There was a 33% reduction in N<sub>2</sub>O and N<sub>2</sub> emissions for Johnstown and a 27% reduction in leached N losses. There was a larger reduction in N<sub>2</sub>O and N<sub>2</sub> emissions (41%) for Solohead due the higher %clay content (and hence higher de-nitrification potential). Leached N losses were reduced by 22%. It should be noted that these modelled reductions were 50% lower than the observed reductions (Selbie *et al.*, 2014) where 70% reductions in N<sub>2</sub>O and 50% reductions in leached N were observed on a free draining cambisol.

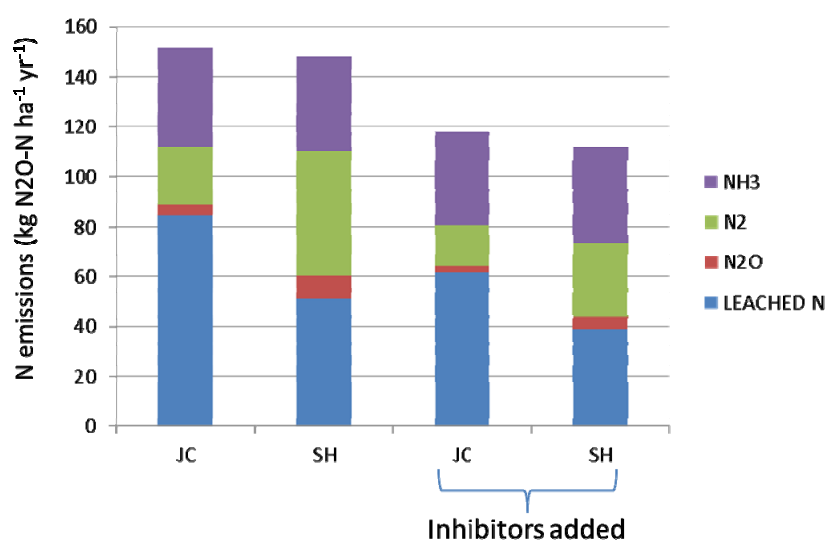


Figure 3.10. Modelled effects of N inhibitors on N emissions from Johnstown Castle (JC) and Solohead (SH). Stocking rates were 2.2 LSU ha<sup>-1</sup> and application of 220 kg N ha<sup>-1</sup>. Two inhibitor applications were simulated and inhibitors were applied in Spring (March) and Autumn (September).

### 3.6. Deficits in modelling of pasture, paddock and range (DNDC)

Modelled outputs of urine deposition (pasture, paddock and range emissions) give similar values of N<sub>2</sub>O loss and N<sub>2</sub>/N<sub>2</sub>O modelled losses were robust for mineral fertiliser. However, N<sub>2</sub> losses were grossly underestimated for urine deposition. Modelled N<sub>2</sub>/N<sub>2</sub>O ratios at an application rate of 700 kg N ha<sup>-1</sup> were 6.5. However, measured N<sub>2</sub>/N<sub>2</sub>O for Solohead and Johnstown was 12 and 50 respectively. The large discrepancy for Johnstown was due to the fact that most of the measured losses (160 kg N) using <sup>15</sup>N tracing was due to co-denitrification, a process not simulated in DNDC.

### 3.7. Summary

Results are shown in Table 3.2 on the next page. The emission factors calculated with the process-oriented model DNDC clearly differed from the IPCC emission factors. Nitrification inhibitors halved  $N_2O$  emission which is not accounted for when IPCC emission factors are applied. Reduced stocking rate, increasing the proportion of clover and applying additional slurry all had profound effects on the simulated emission factors. Simulated factors strongly varied, and were on average lower than IPCC emission factors for several options. Only the average of the emission factors simulated compares to the IPCC emission factor. For the extremes in the whole range of model inputs tested with the individual options differences were profound (Table 3.2).

Modelling with DNDC provides further insight in the effect of several management options at the field level, and more importantly, in dependency of the precise conditions and management factors in place, on the variation to be expected for site-specific emission factors. Explaining variation is prerequisite to evaluate mitigation and adaptation options and their effect on  $N_2O$  emissions in an integrated manner with other sources and sinks of GHG emission, and to identify possible trade-offs between various sources and sinks of GHG emission and system production indicators.

Modelling with PaSim allowed to study the impact of climatic conditions on grassland utilization by livestock, and on GHG emissions by grassland-livestock systems. Effects on  $N_2O$  emission and soil carbon sequestration were tested under a variety of scenarios and locations in France. Results demonstrated a large impact of climate scenarios, of management options and of initial soil / grassland conditions or local conditions on simulated GHG emissions. Although having a different modelling scope, simulation with PaSim and DNDC were very similar on several options:

- 1) an exponential increase of  $N_2O$  emission with increased N fertilisation rate (although difficulty remains in addressing the impact of aridity on soil water content and hence the denitrification process);
- 2) an increased  $N_2O$  emission and decreased C sequestration with increase of the proportion of legumes above a level of 30% (higher emission under arid conditions) and an increased stocking density;
- 3) an increased C sequestration with higher artificial N fertilisation and with application of animal manure.

*Table 3.2 Summary of results for various options tested with the DNDC model*

Description of the option	IPCC emission factor	Range within literature	Main source of variability	Main source of uncertainty	Change of C sequestration (tonnes of C ha <sup>-1</sup> yr <sup>-1</sup> )	Emission factor simulated-N <sub>2</sub> O	Emission factor simulated-CH <sub>4</sub>	Ammonia
Baseline: 2.9 LSU ha <sup>-1</sup> , 226 kg N ha <sup>-1</sup> , total N input = 458 kg N ha <sup>-1</sup>	N <sub>2</sub> O: 2% (PPR) 1% organic manure 1% mineral fertiliser soil C: 0.14 t C ha <sup>-1</sup> yr <sup>-1</sup>	0.18 - 6% (PPR), 0.5 - 6.5% organic manure, 0.4% - 6% mineral fertiliser	N <sub>2</sub> O : soil moisture (precipitation x soil texture) CO <sub>2</sub> : climate (temp x precipitation)	Proportion of N <sub>2</sub> /N <sub>2</sub> O for N <sub>2</sub> O: land-use history for CO <sub>2</sub>	0.25 – 0.5 t C ha <sup>-1</sup> yr <sup>-1</sup> (net biome productivity)	0.9% - 2.5% (global emission factor)	Sink 1.25 kg CH <sub>4</sub> -C ha <sup>-1</sup> yr <sup>-1</sup>	39.8-48.4 kg NH <sub>3</sub> -N ha <sup>-1</sup> yr <sup>-1</sup>
Reduction in stocking rate (by 1 LSU ha <sup>-1</sup> ) fertiliser input 156 kg N ha <sup>-1</sup> . Total N input per hectare = 308 kg N ha <sup>-1</sup>	N <sub>2</sub> O: 2% (PPR) 1% organic manure 1% mineral fertiliser	0.18 - 6% (PPR), 0.5 - 4.5% organic manure, 0.4% - 4% mineral fertiliser	Urine deposition rate and urine composition for N <sub>2</sub> O	C offtake during grazing and C deposition in faeces	Decrease in NBP by 0.08 (Solohead) and 0.1 t C ha <sup>-1</sup> yr <sup>-1</sup> (Johnstown)	0.8% - 1.4%	Decrease in sink capacity 0.13 kg CH <sub>4</sub> -C ha <sup>-1</sup> yr <sup>-1</sup>	Decrease 10 - 14 kg NH <sub>3</sub> -N ha <sup>-1</sup> yr <sup>-1</sup>
Clover addition, slurry (30 t ha <sup>-1</sup> ) no additional fertiliser 1.9 LSU ha <sup>-1</sup> total N input 305 kg N ha <sup>-1</sup> = 66 kg N slurry (39 kg TAN), 80 kg N from fixation,	0% (N <sub>2</sub> O)	0.0 - 0.05% N <sub>2</sub> O	Clover proportion in sward	N fixation rate	No change (if oversown)	0.6- 1.1%	sink 1.18 kg CH <sub>4</sub> -C ha <sup>-1</sup> yr <sup>-1</sup>	Decrease of 15.4 - 19.2 kg NH <sub>3</sub> -N ha <sup>-1</sup> if 60 kg urea is not spread
Addition of extra 60t slurry (90 kg TAN) Note: mineral fertiliser reduced	1% (N <sub>2</sub> O)		Slurry Dry Matter and total ammoniacal N content	Mineralisation rate of slurry C, mineralisation of organic N	Increase by 0.31 t C ha <sup>-1</sup>	0.54% - 1.1%	CH <sub>4</sub> Source 0.21 kg CH <sub>4</sub> -C ha <sup>-1</sup> yr <sup>-1</sup>	Increase of 24.3-41.8 kg NH <sub>3</sub> -N ha <sup>-1</sup> yr <sup>-1</sup>
Nitrification inhibitors	Not in IPCC inventories - 40% reduction in direct/indirect emissions in NZ inventory	33-40% reduction in N <sub>2</sub> O	Rate of nitrification inhibition	Breakdown rate of DCD in soil	No change	0.5 - 1.8% (approx. 30% reduction)	No change	No change

## 4. Individual options at the animal level

At the animal level, individual mitigation options were studied with the Dutch Tier 3 model for enteric fermentation in dairy cows. Mitigation options were in accordance with the outcome of initial discussions with representatives of several work packages in AnimalChange, which relate to various options for farms to adapt to climatic changes. Results are discussed in this chapter and summarised in Table 4.1.

### 4.1. Varying quality of grass silage

The mitigation option improving forage quality via the grass silage diet can be reached by changes in relation to N fertilisation rate and sward weight or stage of grass maturity at moment of cutting. An extreme range of N fertilisation rates of grassland and moment of first cut was simulated to have a strong impact on  $\text{CH}_4 \text{ kg}^{-1}$  DM grass silage, but in particular on  $\text{CH}_4 \text{ kg}^{-1}$  milk produced (from hereon, milk is considered to be fat- and protein- corrected milk). For the latter, differences mounted up to 15% with the lowest  $\text{CH}_4$  emission realized for a highly fertilised grassland and early cut grass, whereas they stayed within 5% difference when expressed per kg grass DM (Figure 4.1). These simulation results demonstrate the principal of the effect of changes in chemical composition and rumen degradability of grass on the direction of changes in  $\text{CH}_4$  emission. For practical conditions far less extreme changes in N fertilisation rate and moment of cutting are feasible. Nevertheless, just a quarter of the size of these measures tested here ( $75 \text{ kg N ha}^{-1}$  higher fertilisation rate;  $375 \text{ kg DM ha}^{-1}$  lower grass harvest for the first cut) still results in an expected 5% lower  $\text{CH}_4$  emission.

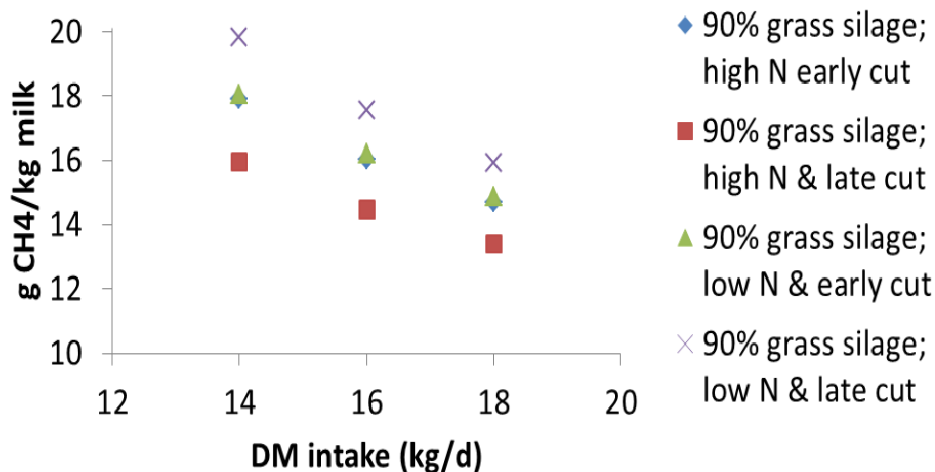


Figure 4.1. Effect of level of DM intake and grass quality on enteric  $\text{CH}_4$  emission simulated with the Dutch Tier 3 for enteric  $\text{CH}_4$  emission in dairy cows. The DM intake ranged from 14 to 18  $\text{kg DM d}^{-1}$  and the diet consisted of 90% grass silage and 10% concentrate on DM basis. Grass silages differed in moment of cutting (early versus late; i.e. a cut of 3.0 versus 4.5  $\text{ton DM ha}^{-1}$ ) and level of N fertilisation (i.e. low versus high; i.e. 150 versus 350  $\text{kg artificial fertiliser N ha}^{-1}$  preceding this cut).

## 4.2. Varying quality of maize silage

The mitigation option improving forage quality via the maize / grass silage diet can be reached by changes in relation to stage of ripening and moment of maize harvest. Extremes in the moment of maize cutting resulted in only a 5% difference in the amount of CH<sub>4</sub> produced per kg of dietary DM, and a 3 to 10% difference per kg of milk produced, depending on whether aminogenic nutrients are protein deficient for optimal rumen microbial activity and or for milk protein synthesis (Figure 4.2).

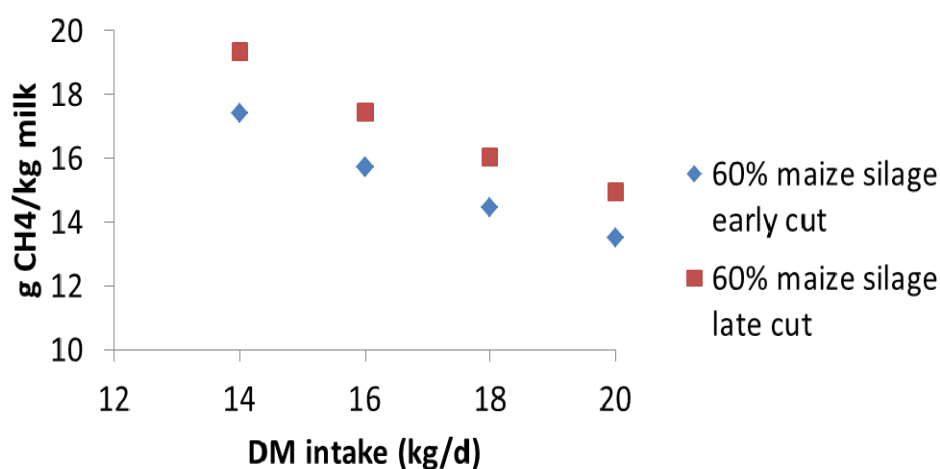


Figure 4.2. Effect of level of DM intake and stage of cutting of maize on enteric CH<sub>4</sub> emission simulated with the Dutch Tier 3 for enteric CH<sub>4</sub> emission in dairy cows. The DM intake ranged from 14 to 20 kg DM d<sup>-1</sup> and the diet consisted of 60% maize silage, 30% grass silage and 10% concentrate on DM basis.

## 4.3. Exchange of grass and maize silage

Exchange of forages (grass silage versus maize silage) may also be a mitigation option. Small effects on CH<sub>4</sub> kg<sup>-1</sup> DM were simulated for the exchange of maize silage for grass silage. When expressed per kg of milk simulated, there was a substantial reduction of 10% in the amount of CH<sub>4</sub> with an increase of the proportion of maize silage up to 30% of dietary DM (Figure 4.3). A further increase of the proportion of maize silage did not show such a decreased CH<sub>4</sub> yield. In the simulations performed the diet was not supplemented with crude protein and with the further increase in the proportion of maize silage above 30% of dietary DM the supply of aminogenic nutrients became limiting for milk production. This limitation reduced milk production and hence limited a further decrease in CH<sub>4</sub> kg<sup>-1</sup> milk with further increase of maize silage proportion in the diet.



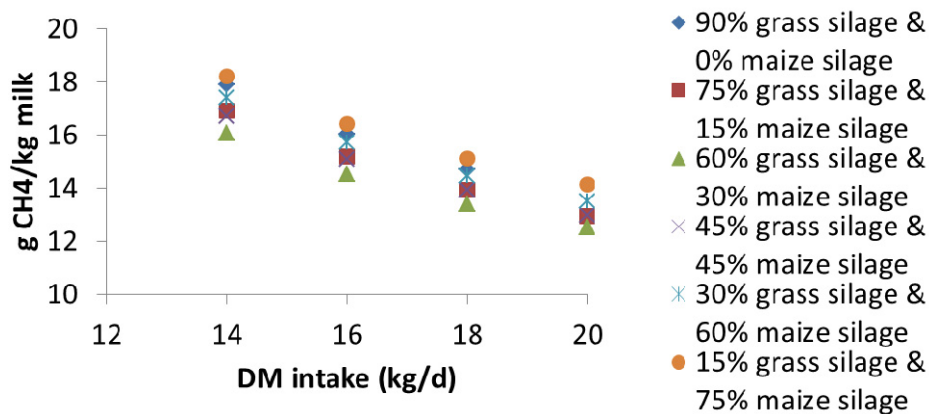


Figure 4.3. Effect of level of DM intake and exchange of grass silage with maize silage on enteric CH<sub>4</sub> emission simulated with the Dutch Tier 3 for enteric CH<sub>4</sub> emission in dairy cows. The DM intake was ranging from 14 to 20 kg DM d<sup>-1</sup> and the diet consisted of 90% forages and 10% concentrate on DM basis.

#### 4.4. Supplementing with various carbohydrate sources

Varying carbohydrates as supplement of grass silage based diet (sugar-rich and starch-rich products, and maize silage) is a mitigation option. Simulated CH<sub>4</sub> kg<sup>-1</sup> DM was highest when a low N, early cut grass silage was supplemented with molasses and wheat, and lowest when supplemented with maize and maize silage. The supplement composed a 30% of dietary DM and the predicted CH<sub>4</sub> production differed by 5%. When expressed per kg of milk produced (Figure 4.4) the supplementation with molasses showed about an 8% higher CH<sub>4</sub> emission compared to the other carbohydrate sources, whereas the most glucogenic nutrients delivering carbohydrates (wheat and maize) showed the lowest values.

#### 4.5. Protein supplementation of grass silage based diets

Protein supplementation of (low N) grass silage based diet (formaldehyde treated soybean meal, untreated soybean meal, high N grass silage, high N grass herbage) is a mitigation option. Effects of DM intake and effects on milk production were included. Simulation of supplementing a low N, early cut grass silage with various protein sources for 20% of dietary DM revealed that a higher CH<sub>4</sub> emission kg<sup>-1</sup> DM was predicted with untreated soybean meal (highly digestible in the rumen) as a protein supplement compared to protein supplementation by treated soybean meal and high N grass silage. Differences remained small however, less than 3%.

When expressed per kg of milk produced (Figure 4.5) the lowest CH<sub>4</sub> emission was predicted for treated and untreated soybean meal and the highest for high N grass silage or grass herbage as protein supplement, with a maximum difference of 20%. Formaldehyde-treated soybean meal demonstrated a 5% lower CH<sub>4</sub> emission per kg of milk compared to untreated soybean meal reflecting its resistance against rumen degradation and lower contribution to rumen fermentable substrate.

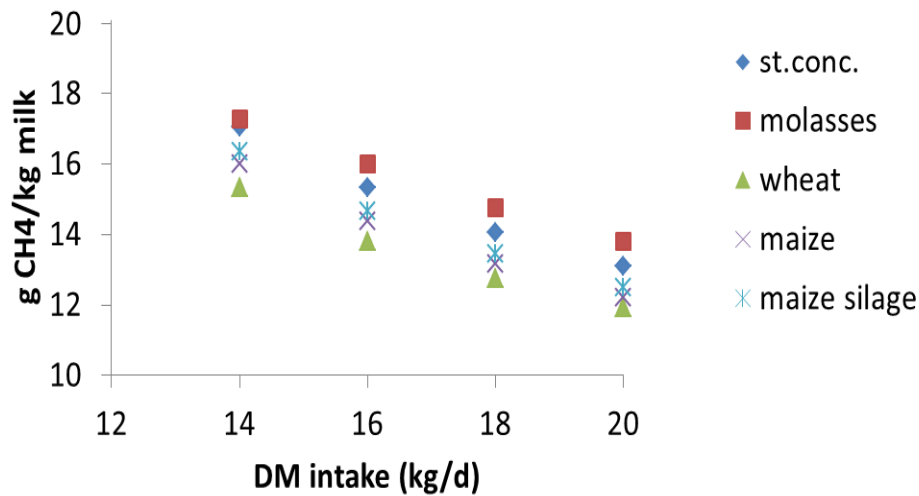


Figure 4.4. Effect of level of DM intake and supplementation with various carbohydrate sources (30% dietary DM; molasses, wheat, maize or maize silage) of a grass silage based diet (70% dietary DM) on enteric CH<sub>4</sub> emission, simulated with the Dutch Tier 3 for enteric CH<sub>4</sub> emission in dairy cows. The DM intake was ranging from 14 to 20 kg DM d<sup>-1</sup> and silage was assumed to be attained with high N fertilisation rate and early cutting.

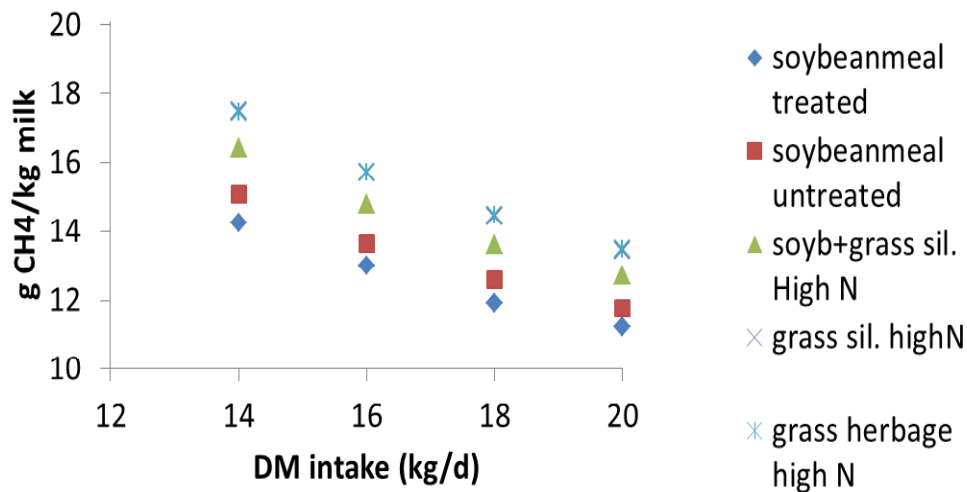


Figure 4.5. Effect of level of DM intake and supplementation with various protein sources of a grass silage based diet on enteric CH<sub>4</sub> emission simulated with the Dutch Tier 3 for enteric CH<sub>4</sub> emission in dairy cows. The DM intake was ranging from 14 to 20 kg DM d<sup>-1</sup>, silage was assumed to be attained with a low N fertilisation rate and early cutting, and the diet was composed of 70% grass silage, 20% protein supplement (treat or untreated soybean meal, soybean meal & high N fertilisation grass silage, high N fertilisation grass silage, and high N fertilisation grass herbage) and 10% concentrates on DM basis.

## 4.6. Protein supplementation of maize silage based diets

Protein supplementation of maize silage based diet (low N grass silage with urea, low N grass silage without urea, untreated soybean meal, high N grass silage) is a mitigation option. Effects of DM intake and effects on milk production were included. Supplementation of a maize silage diet with a protein source diet for 20% of dietary DM was simulated to deliver the highest  $\text{CH}_4 \text{ kg}^{-1} \text{ DM}$  with untreated soybean meal and high N grass herbage as a supplement because of their higher rumen degradability. Differences with high N grass silage (with or without urea) and maize, and with soybean meal as supplement remained small and within 3%.

When expressed in  $\text{CH}_4$  per kg of milk produced (Figure 4.6), soybean meal, maize and soybean meal, and low N grass silage with urea as protein supplement showed about 15% lower  $\text{CH}_4$  emission than for high N grass herbage, high N grass silage, and low N grass silage without urea.

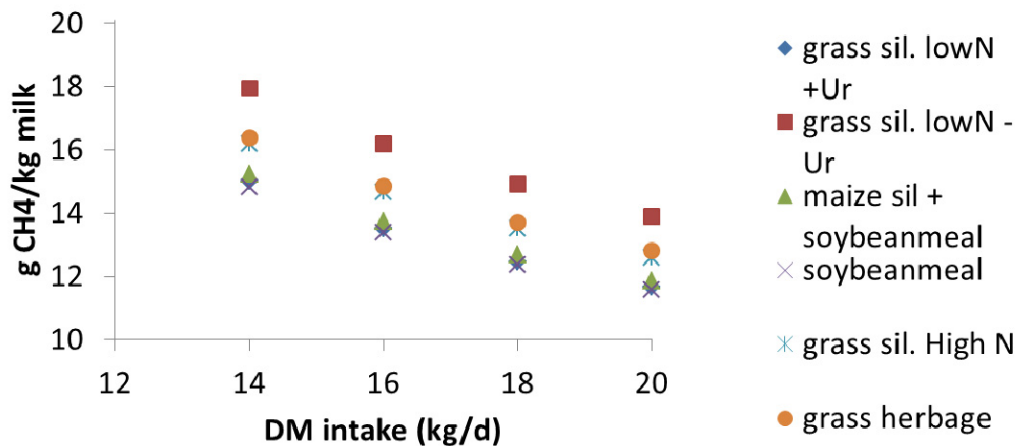


Figure 4.6. Effect of level of DM intake and supplementation with various protein sources of a maize silage based diet on enteric  $\text{CH}_4$  emission simulated with the Dutch Tier 3 for enteric  $\text{CH}_4$  emission in dairy cows. The DM intake was ranging from 14 to 20  $\text{kg DM d}^{-1}$ , the diet was composed of 70% maize silage, 20% protein supplement (low N fertilisation grass silage and urea, low N fertilisation grass silage, maize silage & soybean meal, soybean meal, high N fertilisation grass silage, grass herbage) and 10% concentrates on DM basis.

## 4.7. Fat supplementation and nitrate as methane-reducing additive

The effect of fat supplementation and nitrate as  $\text{CH}_4$  reducing additive is indicated based on results published in literature. Supplementing fat is a very potent measure to reduce  $\text{CH}_4$  emission in cows, if dietary levels are kept below threshold levels. With every 1% of increase of the fat content of dietary DM the  $\text{CH}_4$  emission reduces with  $1 \text{ g CH}_4 \text{ kg DM}^{-1}$  which is a 5% decrease when the basal diet (excluding the fat source) would deliver  $20 \text{ g CH}_4 \text{ kg DM}^{-1}$ . A 4% of dietary DM as supplemented fat would reduce  $\text{CH}_4$  emission by 20%. Although fat supplementation is a very potent measure to reduce  $\text{CH}_4$ , this measure may not be feasible during the whole lactation cycle, and important trade-offs may be a reduced digestibility of the fibrous part of the diet as well as a reduced feed intake.

Another potent measure to mitigate CH<sub>4</sub> is the addition of nitrate. Addition of nitrate at 0.5% of dietary DM reduces CH<sub>4</sub> emission by 0.8 g kg DM<sup>-1</sup> which is 4% when the basal diet would deliver 20 g CH<sub>4</sub> kg DM<sup>-1</sup>. An addition of nitrate at a level of 1% of dietary DM may reduce CH<sub>4</sub> emission with a maximum of 10%. With a nitrate level of 2% of dietary DM a persistent reduction in CH<sub>4</sub> emission by 16% has been measured in dairy cows by Van Zijderveld *et al.* (2011).

## **4.8. Additional factors to consider when comparing feeding measures**

### **4.8.1. Effect of feeding measures on feed intake**

The relative differences between individual feeding measures in their effect on CH<sub>4</sub> emission remained rather consistent across a level of feed intake ranging from 14 to 20 kg DM d<sup>-1</sup>, which would cover average feed intake established by the average dairy cow in various production conditions. Feed intake level in itself had a higher impact on CH<sub>4</sub> kg<sup>-1</sup> DM or CH<sub>4</sub> kg<sup>-1</sup> milk than the feeding measures evaluated. Feed intake ranging from 14 to 20 kg DM intake d<sup>-1</sup> caused roughly 20% differences for most of the diets and measures simulated. This means that not only the effect of a feeding measure in itself on enteric fermentation needs to be evaluated, but also the accompanying effect of that measure on feed intake level achieved.

The present study does not give an indication of such effects on feed intake however. The process-oriented model does not include predictions of (changes in) feed intake, but requires this as input. It is difficult to predict effects on feed intake, but estimates may be derived from trials reported in literature, or from insights in practice or from models that have been developed to evaluate feed intake effects.

### **4.8.2. Effect of feeding measures on milk yield**

Effects on milk yield may be calculated from the intake of metabolizable energy or net energy of lactation. However, model calculations show for several feeding scenarios that predicted milk yield may be limited by the supply of aminogenic or glucogenic nutrients, and not by energy supply. A lower milk yield than the potential yield expected based on energy supply occurred in particular with 1) low N grass silage diets with a limiting glucogenic nutrient supply when starch-rich carbohydrates or maize silage is lacking, 2) maize silage diets which lack a protein supplementation. Nutrient limitation of milk yield may hence strongly affect the effect of a feeding measure on CH<sub>4</sub> kg<sup>-1</sup> milk. Simulation results of the present study indicate that in some cases such effects on milk production may be of a similar magnitude than the simulated effect of the feeding measures in itself.

Results show that, next to the effect of a feeding measure on the level of feed intake, the supply of aminogenic and glucogenic nutrients and their potential limitation of milk production is a further aspect to be taken into account when evaluating effects of feeding measures on enteric CH<sub>4</sub> emission and cow productivity. When diets become well balanced for glucogenic and aminogenic nutrient supply, the size of effects on CH<sub>4</sub> kg<sup>-1</sup> milk may become smaller than simulated here.

## 4.9. Introducing clover in grassland

The proportion of clover in grass swards is associated with levels of N fertilisation. For a selection of the cases evaluated with the DNDC model as described in 3.2, the enteric fermentation model was used to predict the consequences for enteric CH<sub>4</sub>. Observations of chemical composition and of digestibility of the grass (6% clover) and the grass/clover sward (22% clover) remained very similar. As a result, model inputs hardly differed and reported DM intake was similar, leading to similar predictions of enteric CH<sub>4</sub> emission (results not shown).

Under other conditions the effects N fertilisation rates, stocking densities, and climatic conditions may have more pronounced impact on herbage quality and intake, which is expected to lead to larger differences in CH<sub>4</sub> emission. However, for the present case studies a change from grass to grass/clover does not seem have a large effect on enteric CH<sub>4</sub> emission. This result corresponds with that reported for PASIM which will be discussed in Chapter 5.

## 4.10. Varying stocking density and grazing time

Stocking density and restricted grazing time affect CH<sub>4</sub> emissions in grazed systems. For a selected case study where very high stocking densities were applied (as opposed to the much lower densities studied in 3.4 and discussed for the Irish farm case in Chapter 6) a lower enteric CH<sub>4</sub> kg<sup>-1</sup> milk was estimated with lower stocking density (results not shown). This was caused by the higher herbage allowance and intake by cows with a reduced stocking density, resulting in a 15% and 19% higher milk yield in two consecutive monitoring rounds, and hence lower CH<sub>4</sub> emission kg<sup>-1</sup> milk. An increase of stocking density from 4.5 to 6.4 cows ha<sup>-1</sup> led to an increase in CH<sub>4</sub> kg<sup>-1</sup> milk of 17% and 19% in the two monitoring rounds. Simulated effects of variation in stocking density hence have to be attributed mainly to changes in cow DM intake and milk yield.

Stocking densities from 4.5 to 6.4 are very high for a unrestricted grazing system however. Much lower densities of 2.2 apply to the Irish farm case that will be discussed in Chapter 6 and that already have been shown for DNDC in 3.4. The result illustrate however how level of intensity of farming or grazing influences cow performance and CH<sub>4</sub> emission intensity.

Also the effect of restriction grazing time in an Irish case study (option also discussed with PaSim in 3.3), or a limited accessing time to grass herbage, was simulated with the enteric fermentation model. The stocking density and N fertilisation was equal and daily allowance of herbage remained the same with 15.5 kg DM d<sup>-1</sup> cow<sup>-1</sup> and an grazing access time of 22 h d<sup>-1</sup>, 9 h d<sup>-1</sup>, 2 times 4.5 h d<sup>-1</sup> or 2 times 3 h d<sup>-1</sup>. Despite the extreme differences in access time the measure had very little effect on predicted CH<sub>4</sub> kg<sup>-1</sup> milk. With the 9 h access period the DM intake was 10% lower compared to the access time options, and resulted in a 9% higher CH<sub>4</sub> kg<sup>-1</sup> milk. Otherwise, effects on cow performance and grass characteristics remained small. These results again emphasize that with this type of management options cow performance is likely to have the largest impact on CH<sub>4</sub> emission intensity with milk production.

## 4.11. Summary

Given DM intake for a specific diet under specific farming conditions, the process-oriented Tier 3 model for enteric CH<sub>4</sub> emission provides detailed insight in how the diet causes variation in rumen fermentation and CH<sub>4</sub> emission. As an alternative to dietary energy values based on (estimates of) faecal digestibility with the Tier 2 approach, the Tier 3 approach explains the impact of variation in chemical composition and intrinsic rumen degradation characteristics of the chemical fractions in dietary DM by a mechanistic representation of microbial activity in the rumen and large intestine. This allows prediction of the consequences of variation in quality of forages and effect of dietary supplementation with starch, protein and fat supplements on enteric CH<sub>4</sub>, diet digestibility and cow performance. Explaining such variation is prerequisite for a case-specific evaluation of the effect of mitigation and adaptation options on enteric CH<sub>4</sub> emission, and of the possible trade-offs or synergies with other GHG sources and sinks.

At the animal level there are several mitigation options available to reduce enteric CH<sub>4</sub> kg<sup>-1</sup> milk. In all cases cow productivity, as a result of DM intake, feed digestion (with a main role for rumen fermentation) and milk production, strongly determine intensity of CH<sub>4</sub> emission when expressed as CH<sub>4</sub> kg<sup>-1</sup> DM and CH<sub>4</sub> kg<sup>-1</sup> milk. When expressed as CH<sub>4</sub> ha<sup>-1</sup> the same factors maintain to have this role. Stocking density and intensity of the dairy farm system and dependency on inputs to the farm become more prominent however when comparing dairy farming systems with varying intensity.

Effective measures to mitigate enteric CH<sub>4</sub> emission are a high N fertilisation of grassland, an earlier cutting of grass, exchange of grass silage by maize silage, inclusion of starch-rich supplements in grass-based diets, and inclusion of protein-rich supplements in maize silage-based diets. To a lesser extent also later cutting of maize crop might be a mitigation option. The results are summarised in Table 4.1 and on average are lower than the IPCC Tier 2 default values. The CH<sub>4</sub> conversion factors (CH<sub>4</sub> energy expressed as % of gross energy intake) differ from the IPCC Tier 2 default with many options. Other nutritional measures that mitigate CH<sub>4</sub> emission with high certainty are the addition of fat and nitrate to the diet. Both do not need to have a detrimental effects on diet digestibility and cow performance up to maximum fat level of 7% of dietary DM, and up to a maximum of nitrate level of 2% of dietary DM (the latter dependent on the method of nitrate allowance to animals and the level of control the farmer can exert on nitrate intake).



Table 4.1 Summary of results simulated with the Dutch Tier 3 for enteric CH<sub>4</sub> in dairy cows, added with results from literature

Description of the dietary options (given in % DM)	IPCC emission factor (CH <sub>4</sub> energy as % of GE intake)	Range within literature (CH <sub>4</sub> energy as % of GE intake)	Main source of variability	Main source of uncertainty	Change of C sequestration (t C ha <sup>-1</sup> yr <sup>-1</sup> )	Emission factor simulated N2O	Emission factor simulated CH <sub>4</sub> (CH <sub>4</sub> energy as % of GE intake)	Urine N simulated as source of ammonia (g d <sup>-1</sup> )	Model used
Changing quality of grass silage with 90% grass silage 10% concentrate (N fertilisation rate and sward weight at cutting)	6.5% default; measure not in IPCC inventories	5.5 – 7.0%	DM intake, grass composition (protein, sugar, NDF), rumen digestion	Rumen degradability NDF, rumen fermentation profile			5.4 - 6.5% of GE intake	133-455 g urine N d <sup>-1</sup>	Dutch Tier 3
Changing quality maize silage with 60% maize silage 30% grass silage 10% concentrate (early vs. late cutting)	6.5% default; measure not in IPCC inventories	5.5 – 6.5%	DM intake, starch content, rumen (& large intestinal) digestion NDF & starch	Rumen degradability NDF and starch, rumen fermentation profile			5.4 - 6.2% of GE intake	139-177 g urine N d <sup>-1</sup>	Dutch Tier 3
Exchange of forage type with 90% forage 10% concentrate (exchange maize silage and good quality grass silage)	6.5% default; measure not in IPCC inventories	5.9 – 7.0%	DM intake, rumen and total digestion NDF & starch	Rumen degradability starch, NDF and CP, rumen fermentation profile			5.4 - 6.2% of GE intake	71-160 g urine N d <sup>-1</sup>	Dutch Tier 3
Carbohydrates supplement with 70% grass silage, 30% supplement (molasses, general compound feed, wheat, maize, maize silage)	6.5% default; measure not in IPCC inventories	5.7 – 7.0 %	DM intake, rumen digestion NDF & starch	Rumen degradability (part. NDF), rumen fermentation profile			5.6– 6.8% of GE intake		Dutch Tier 3
Protein supplemented with 70% grass silage 10% concentrate 20% supplement (soybean meal treated or untreated, high N grass silage & soybean meal, high N grass silage or	6.5% default; measure not in IPCC inventories	5.6 – 6.8%	DM intake, rumen digestion & microbial activity	Rumen degradability NDF, rumen fermentation profile			5.7- 6.3% of GE intake		Dutch Tier 3



<i>grass herbage</i> )									
Protein supplement with 70% maize silage 10% concentrate 20% supplement ( <i>low N grass silage, urea, maize silage &amp; soybean meal, soybean meal, high N grass silage or herbage</i> )	6.5% default; measure not in IPCC inventories		DM intake, rumen digestion & microbial activity	Rumen degradability NDF & starch, rumen fermentation profile			5.5- 6.3% of GE intake		Dutch Tier 3
Fat supplementation	6.5% default; measure not in IPCC inventories	5% reduction of default of 6.5% GE intake per 1% increase of fat in dietary DM (up to max fat content of 10% DM)	Negative effects on DM intake & rumen digestion, level of protection to prevent effects on rumen fermentation	Rumen NDF degradability, (rumen fermentation profile)			Depends on fat dose and basal ration  Fat content >7% than account of effects on intake and digestibility	Reduced by fat dilution of dietary N	Energy aspects covered by Dutch Tier 3  WP6-Animal Change  Literature
Nitrate supplementation	6.5% default; measure not in IPCC inventories	5% reduction of default of 6.5% GE intake per 0.5% of nitrate in dietary DM (depends on DM intake; efficacy of 80% assumed; max. 2% nitrate in dietary DM)	Dosage & efficacy rumen nitrate reduction	Rate of nitrate reduction & nitrate/nitrite absorption or outflow, rumen fermentation profile, (health issues DM intake around max dosage)			Depends on nitrate dose and basal ration  Nitrate to be safely fed up to 2% of dietary DM	Neutral with dietary urea exchanged,  Increased when added to diet without urea exchange	Not covered by Dutch Tier 3  WP6-Animal Change  Literature

## 5. Climate effects on mitigation options: case studies from European grasslands

### 5.1. Introduction

Grassland-based production systems result in three major GHG emissions - CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub> - with fluxes closely linked with management practices, soil types and climatic conditions (Soussana *et al.*, 2004). Soil N<sub>2</sub>O emissions result from microbial nitrate reduction (denitrification) and oxidation (nitrification) and are enhanced by N fertilisation, atmospheric N deposition and biological N fixation by legumes (Mosier *et al.*, 1998). The magnitude of N<sub>2</sub>O 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, CH<sub>4</sub> emissions are dominated by enteric fermentation in ruminants and emissions from their effluents. Ruminant animals release approximately 5% of the ingested digestible C as CH<sub>4</sub> (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 under a range of grassland management 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 CO<sub>2</sub> eq. per year by 2030 of which approximately 1.5 Gt CO<sub>2</sub> 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 can, therefore, be partly mitigated by grassland C sequestration in soil organic matter (Soussana *et al.*, 2010b).

With the purpose of assessing mitigation options, a modelling exercise was performed at grassland sites representative of conditions from Northern, Central and Southern Europe, in which agricultural management options were manipulated (extensification vs. intensification) and their impact on the three main GHG emissions was assessed.

### 5.2. Study sites

Three semi-natural grassland sites were selected covering a gradient of geographic and climatic conditions in Europe (Table 5.1), as well as a variety of management practices (Table 5.2) and soil types (Table 5.3).

The three sites are representative of Northern (Easter Bush, United Kingdom; Soussana *et al.*, 2007), Central (Laqueuille, France; Klumpp *et al.*, 2011) and Southern (Val D'Alinyà, Spain; Wohlfahrt *et al.*, 2008) Europe, distributed over a gradient of latitudes (about 42° to 56° North) and of elevations up to about 1800 m a.s.l. (Table 5.1). Along these gradients, the mean annual temperature varies from about 6 (Vall d'Alinyà, Spain) to 9 °C (Easter Bush, United Kingdom) with annual precipitation rates around 1000 mm on average. With respect to management (Table 5.2), the dataset include extensively and intensively managed grasslands (in terms of grazing intensity and nitrogen fertilisation), representative of mixed grass swards (in the presence of 12% clover at Laqueuille).

Table 5.1. Location and climate of the grassland study sites.

Country	Site	Years	Altitude (m a.s.l)	Climate		Coordinates	
				Mean air temperature (°C)	Precipitation total (mm yr <sup>-1</sup> )	Lat	Long
France	Laqueuille	2002-2011	1040	7.8	1072	45.63	2.73
United Kingdom	Easter Bush	2002-2008	190	9.0	956	55.86	3.20
Spain	Vall D'Alinyà	2004-2008	1770	6.2	908	42.15	1.44

Table 5.2. Management of the grassland study sites.

Country	Site	Utilization	Management (average)			
			Nitrogen fertilisation		Grazing	
			Events (yr <sup>-1</sup> )	Total amount (kg N ha <sup>-1</sup> yr <sup>-1</sup> )	Stocking rate (LSU ha <sup>-1</sup> yr <sup>-1</sup> )	Duration (d yr <sup>-1</sup> )
France	Laqueuille	Intensive	3	210	1.10	160
United Kingdom	Easter Bush	Extensive	4	200	0.65	160
Spain	Vall D'Alinyà	Extensive	-	-	0.18	120

Table 5.3. Soil properties of the grassland study sites.

Country	Site	Soil properties					
		Soil depth (m)	Soil texture				Bulk density (t ha <sup>-1</sup> )
			Sand (%)	Silt (%)	Clay (%)	Type	
France	Laqueuille	0.7	27	53	20	Silt loam	0.87
United Kingdom	Easter Bush	0.7	12	26	62	Clay	1.47
Spain	Vall D'Alinyà	0.6	15	50	35	Silty clay loam	0.87

Monthly and yearly summaries for the average temperature and total precipitation were obtained from the hourly weather data available for each site. The De Martonne aridity index ( $b$ ) was elaborated from the formula of Gottmann (De Martonne, 1942), which accounts for both yearly and within-year variability of temperature and precipitation. The range limits discriminate between thermo-pluviometric conditions associated with aridity gradients:  $b < 5$ : extreme aridity;  $5 \leq b \leq 14$ : aridity;  $15 \leq b \leq 19$ : semi-aridity;  $20 \leq b \leq 29$ : sub-humidity;  $30 \leq b \leq 59$ : humidity;  $b > 59$ : strong humidity (Diodato and Ceccarelli, 2004). The aridity pattern is characterized by sub-humid to humid conditions (Figure 5.1), the average value of De Martonne-Gottmann index ( $b$ ) being: 28 at Vall d'Alinyà (Spain), 31 at Easter Bush (United Kingdom), and 41 at Laqueuille (France). For the latter, the interannual variability of aridity conditions (from  $b=31$  in 2005 to  $b=48$  in 2007 and 2009) is not great. On the other hand, the other two sites can experience sub-humid ( $b=20$  in 2007 at Vall d'Alinyà) to semi-arid ( $b=19$  in 2003 at Easter Bush) conditions.

Three contrasting years in terms of aridity (humid, median and arid) were selected at each site (based on observed climate data) according to the De Martonne-Gottmann aridity index.

Table 5.4. Selected contrasting years in terms of aridity, based on the De Martonne-Gottmann aridity index ( $b$ ).

Site / Aridity conditions	Humid		Median		Arid	
	Year	$b$	Year	$b$	Year	$b$
Laqueuille	2009	48	2006	42	2005	31
Easter Bush	2002	44	2004	33	2003	19
Vall D'Alinyà	2008	39	2004	29	2007	20

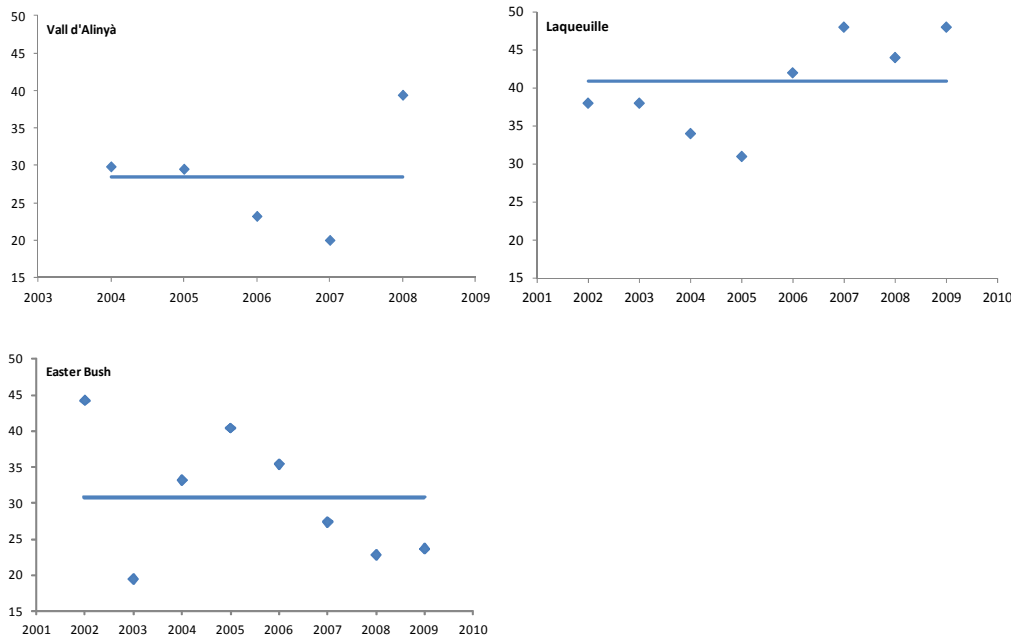


Figure 5.1. Annual values (dots) and average (line) of the aridity index at the grassland study sites.



### 5.3. Simulation study

Three mitigation options - fertilisation rate (with the exclusion of Vall d'Alinyà), legume fraction (only at Laqueuille), animal density - were tested at each site following a factorial approach by increasing/decreasing by 30% the levels of each option, which makes 39 simulations (27 at Laqueuille, nine at Easter Bush and three at Vall d'Alinyà). This was based on a protocol established in the frame of EU-FP7 GHG-Europe (<http://www.ghg-europe.eu>). The decreasing option (-30%) is a real mitigation option, while the alternative choice (+30%) serves the purpose of assessing if an intensification option leads to increased emissions. The effectiveness of mitigation options was assessed with PaSim. The model was parameterized based on the parameterization established by Ben Touhami (2014) for European grassland systems 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 at each site following Lardy *et al.* (2011). PaSim was run at each grassland site to simulate daily values of net ecosystem CO<sub>2</sub> exchanges, NEE (NEE = RECO-GPP, where GPP is gross primary production, RECO is ecosystem respiration), as well as N<sub>2</sub>O and CH<sub>4</sub> emissions. Outputs for the three GHGs were presented as yearly cumulated values. For NEE, positive values indicate the system is a source of C losses, while negative values indicate that the system sequesters C from the atmosphere.

### 5.4. Greenhouse gas emissions

#### ***Vall d'Alinyà***

The results of the effect of livestock intensity on estimated GHG emissions at Vall d'Alinyà are illustrated in Figure 5.2, in which appreciable differences were only observed for CH<sub>4</sub> emissions. When reducing by 30% the animal density, it was estimated about 29% reduction of CH<sub>4</sub> emissions and only 3% of N<sub>2</sub>O releases. For NEE, the system is globally a sink of C, with the exception of arid and intermediate years, while no substantial differences were found for different levels of animal density.



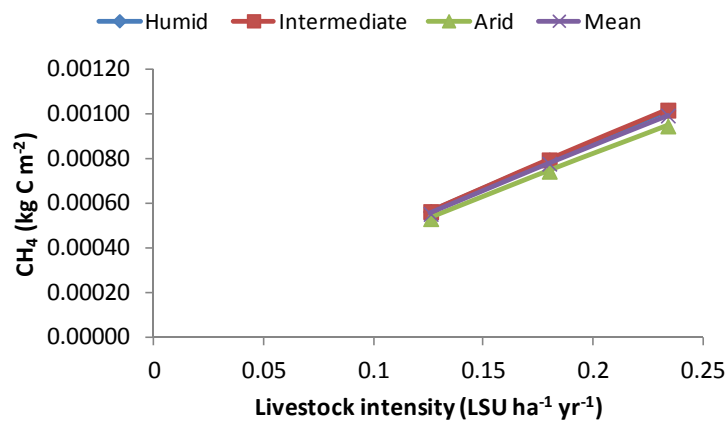
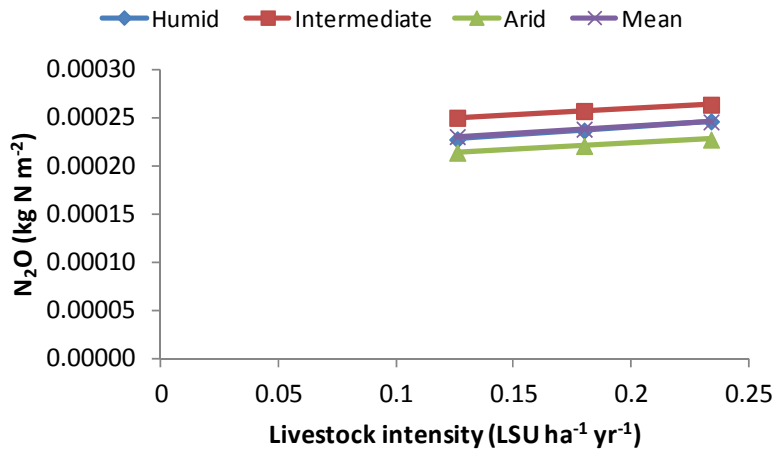
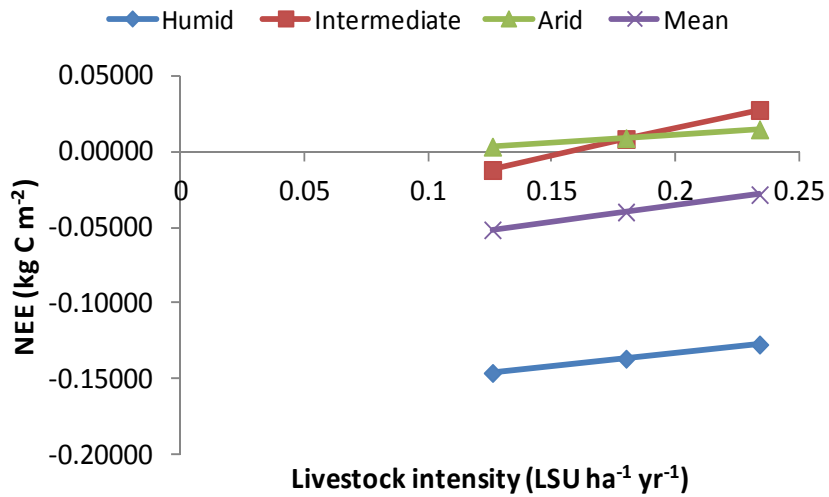


Figure 5.2. Annual GHG emissions (NEE: net ecosystem CO<sub>2</sub> exchanges; NO<sub>2</sub>: nitrous oxide releases; CH<sub>4</sub>: enteric methane releases) simulated by PaSim at Vall d'Alinyà (Spain) for contrasting years (from arid to humid) and for alternative grazing animal intensities.



### Easter Bush

Based on NEE estimates (Figure 5.3), the grassland system in place at Easter Bush is a sink of C with the exception of arid years, where the site tends to become a source of C losses.

Figure 5.4 shows the variations in N<sub>2</sub>O emission estimated over N fertilisation gradient for different intensities of grazing. Globally, emissions are more important in less humid years. The simulation study found that a reduction of about 30% in N fertilisation could be expected to result (on average) in a reduction of about 10% of N<sub>2</sub>O emissions.

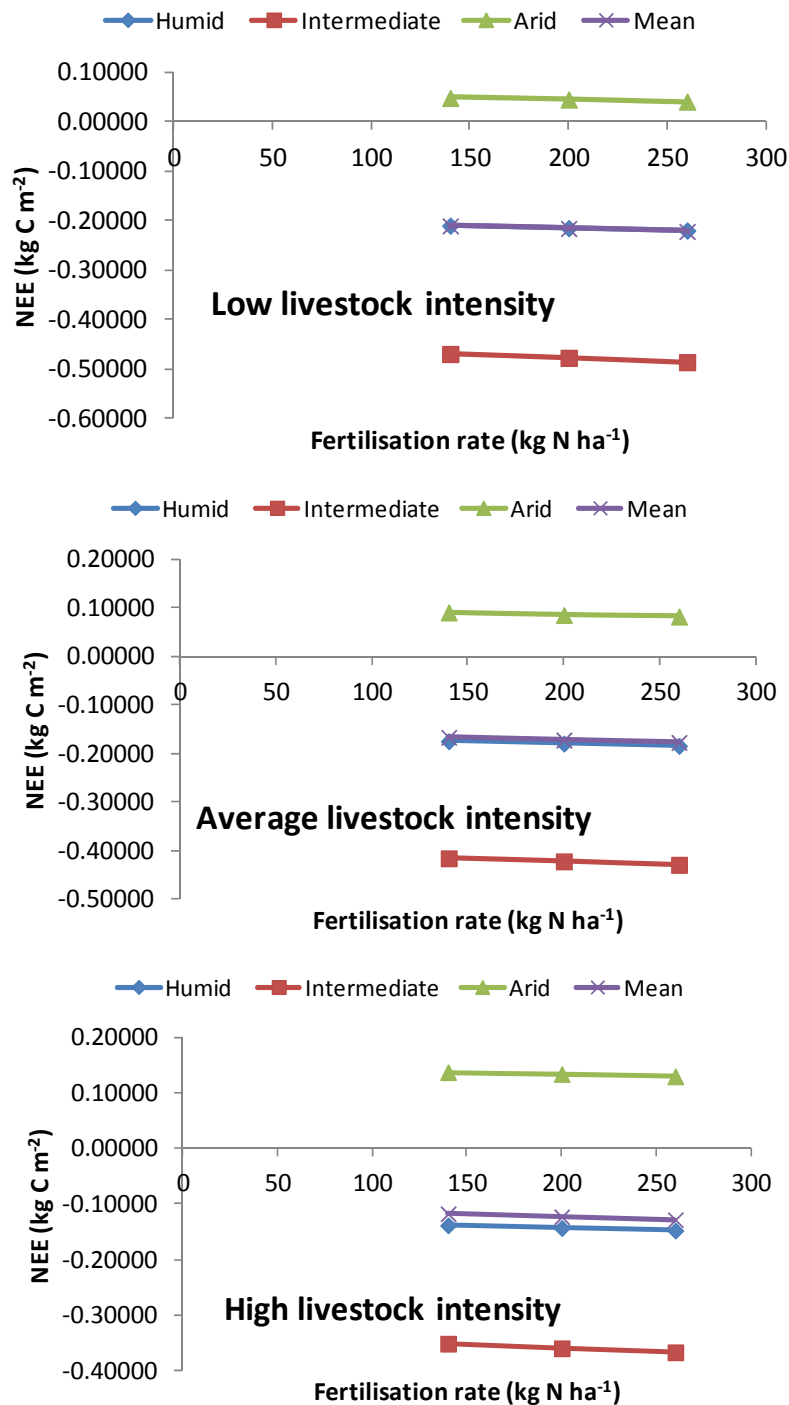


Figure 5.3. Annual net ecosystem CO<sub>2</sub> exchanges (NEE) simulated by PaSim at Easter Bush (United Kingdom) for alternative Nitrogen fertilisation rates and grazing animal intensities.

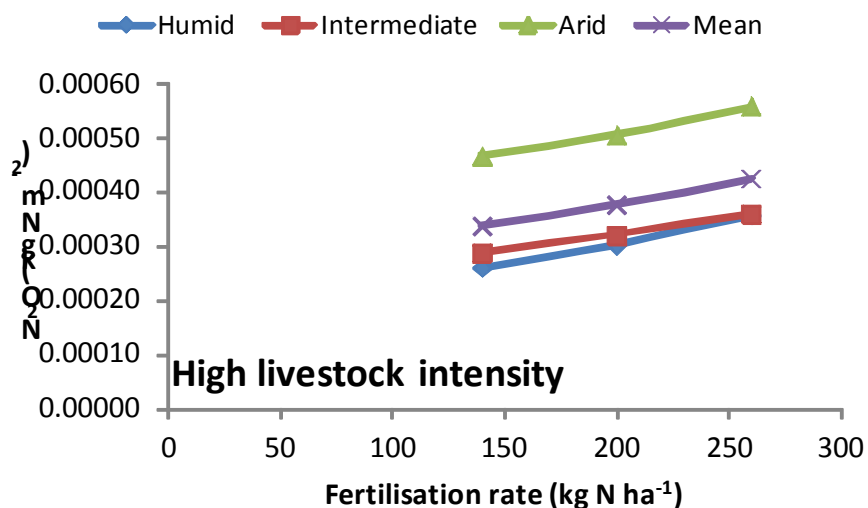
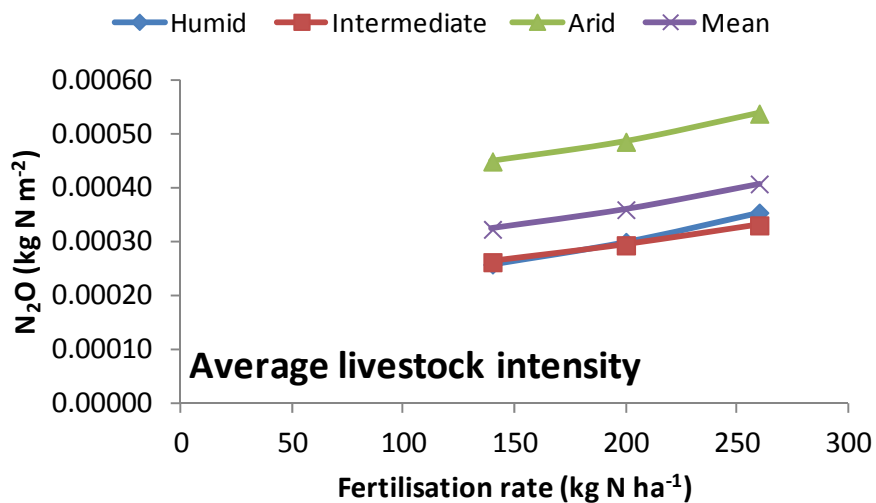
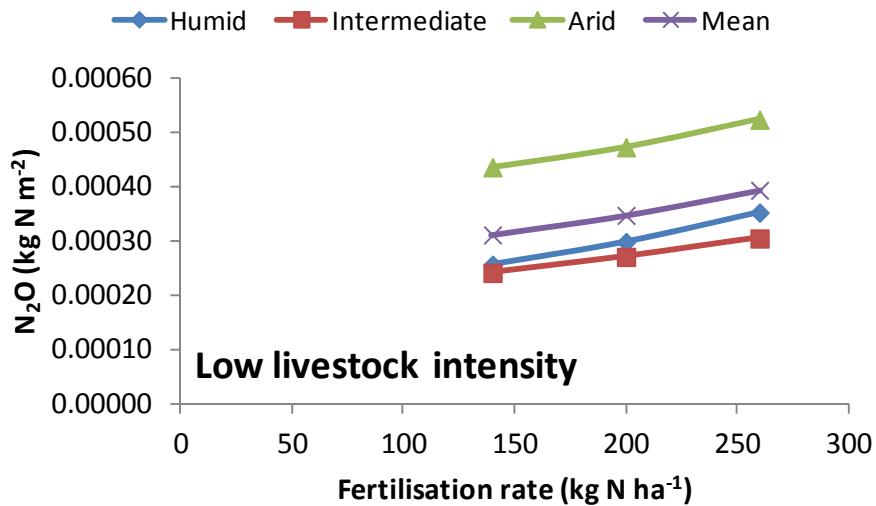


Figure 5.4. Annual nitrous oxide (N<sub>2</sub>O) emissions simulated by PaSim at Easter Bush (United Kingdom) for contrasting years (from arid to humid) and for alternative Nitrogen fertilisation rates and grazing animal intensities.





Figure 5.5 highlights the impact of grazing density on CH<sub>4</sub> emissions for different fertilisation rates. These emissions are expected to decrease (on average) by about 27% with 30% animal density decrease, being more pronounced at intermediate and arid years than humid climate conditions, presumably due to changes in diet quality leading to higher CH<sub>4</sub> emissions under dry weather conditions (e.g. Pinares-Patiño, 2007). During dry years, forage quality declines (e.g. less sugar and crude proteins, more fibre and lignin) due to plant water stress. Forage digestibility is related to the lignin content, since lignin is indigestible by enzymes in ruminant animals. Thus, concurrently with increased lignification, the proportion of organic matter digestibility decreases over time while CH<sub>4</sub> emissions increase.

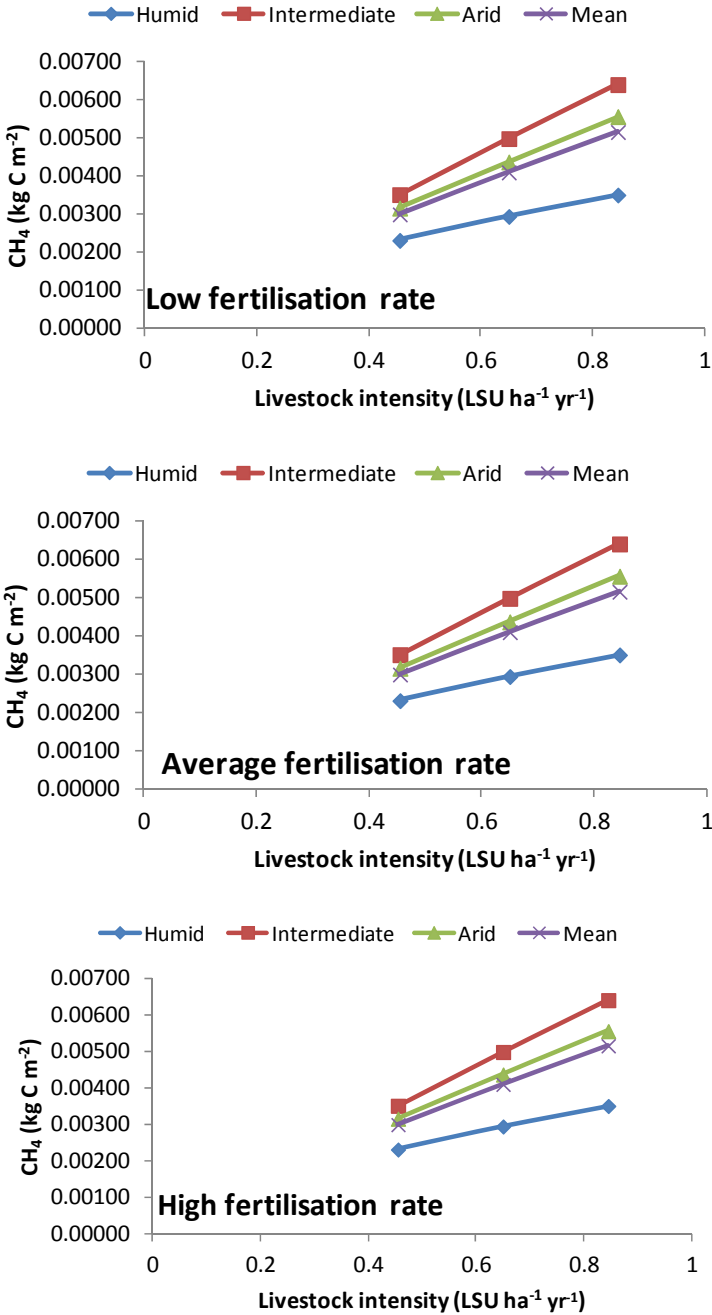


Figure 5.5. Annual enteric methane emission simulated by PaSim at Easter Bush (United Kingdom) for contrasting years (from arid to humid) and for alternative grazing animal intensities and Nitrogen fertilisation rates.



## Laqueuille

Figure 5.6 shows GHG emissions as estimated at Laqueuille from different proportions of leguminous in the sward, with high grazing intensity and fertilisation rate. Overall, the gradient of clover fraction explored was seen not to affect GHG emissions, and this was also true with other combinations of factors (not shown). The system was estimated to be a sink of C with whatever climatic conditions, where drier conditions seem to favour potential C storage (Figure 5.6, top). Under those dry conditions, N<sub>2</sub>O emissions were logically low (Figure 5.6, middle), while CH<sub>4</sub> emissions were high, presumably due to changes in forage quality (Figure 5.6, bottom).

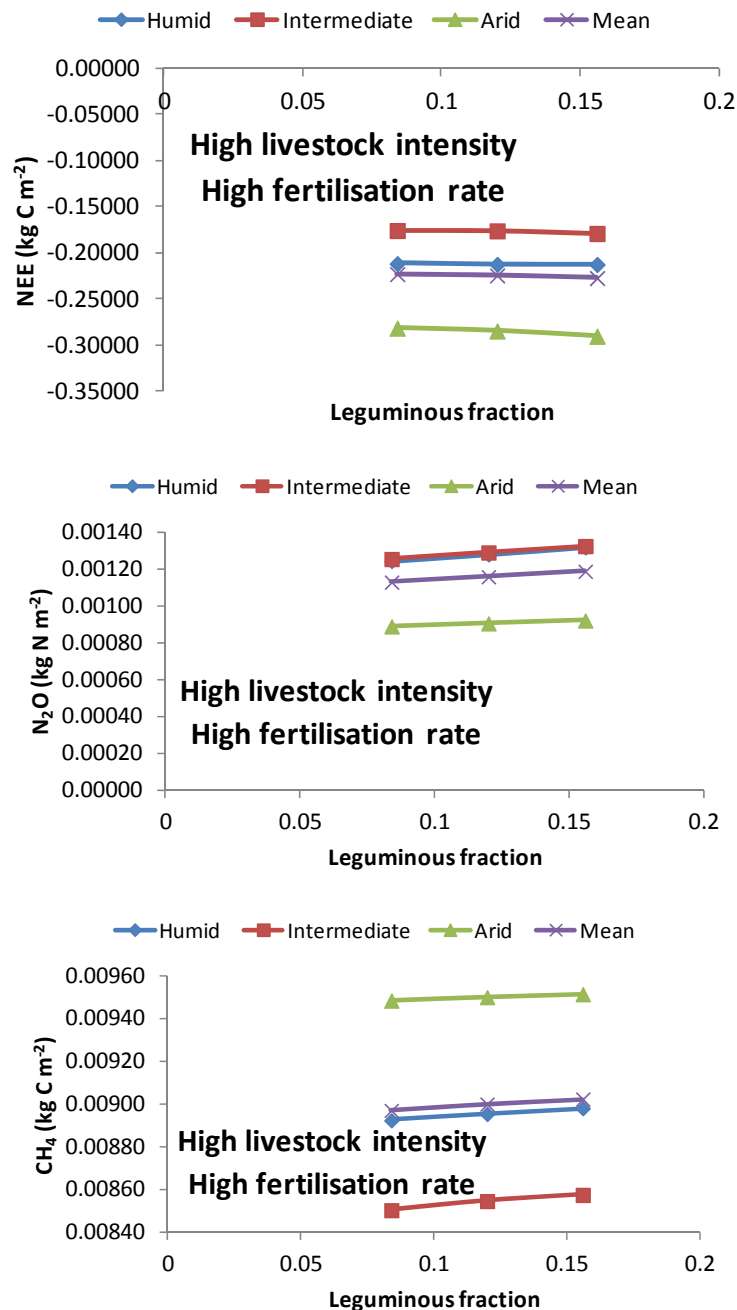


Figure 5.6. Annual GHG emissions (NEE: net ecosystem CO<sub>2</sub> exchanges; NO<sub>2</sub>: nitrous oxide releases; CH<sub>4</sub>: enteric methane releases) simulated by PaSim at Laqueuille (France) for contrasting years (from arid to humid) and for alternative leguminous fractions, grazing animal intensities and Nitrogen fertilisation rates.

Taking the average fraction of leguminous as reference it was found that, with any level of N fertilisation, CH<sub>4</sub> emissions can be reduced by 27-28% with 30% reduction of grazing animals (Figure 5.7). Likewise, independently on livestock density, estimated N<sub>2</sub>O emissions were reduced by about 23% with 30% reduction of N fertilisation rate (Figure 5.8), underlining the strong impact of N supply compared to livestock density. Given the higher incidence of non-CO<sub>2</sub> emissions on the global warming effect, a reduction of N fertilisation should be considered.



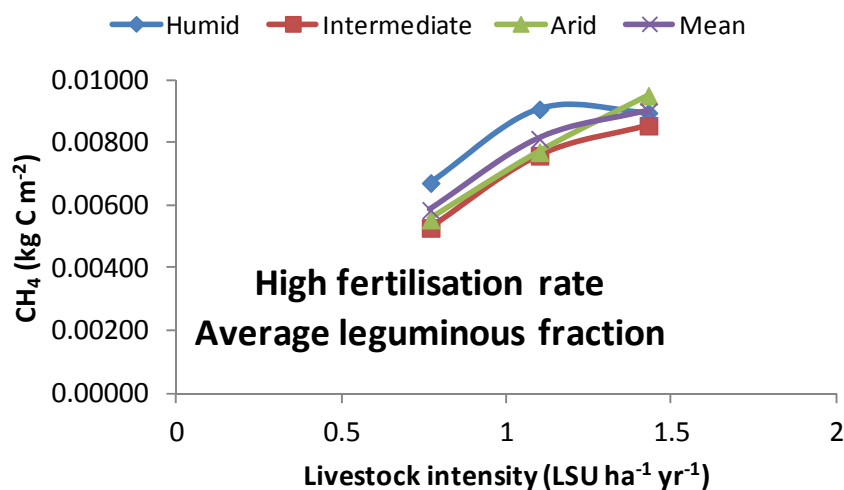
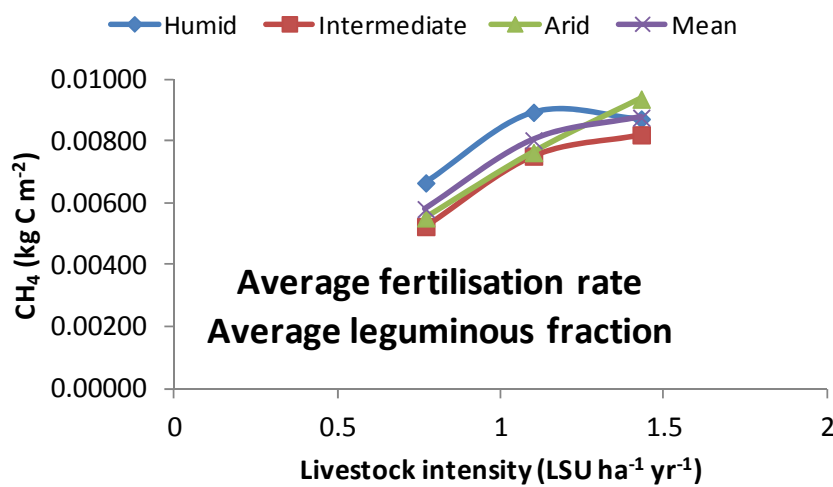
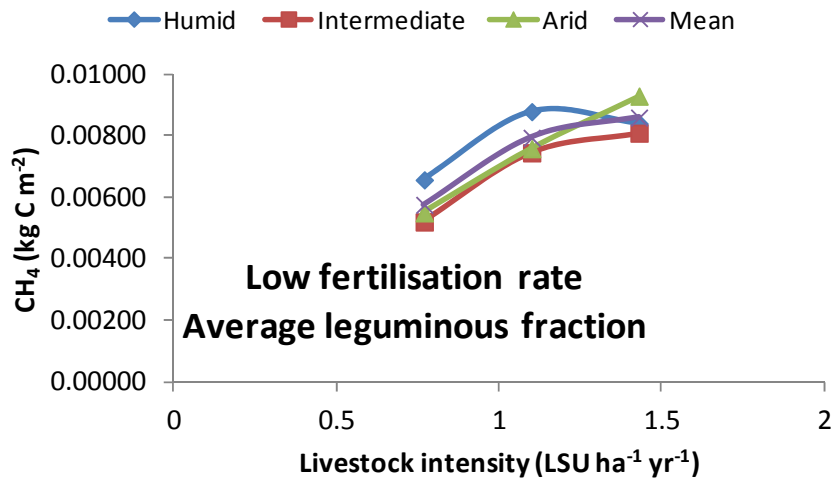


Figure 5.7. Annual enteric methane emission simulated by PaSim at Laqueuille (France) for contrasting years (from arid to humid) and for alternative grazing animal intensities and nitrogen fertilisation rates.



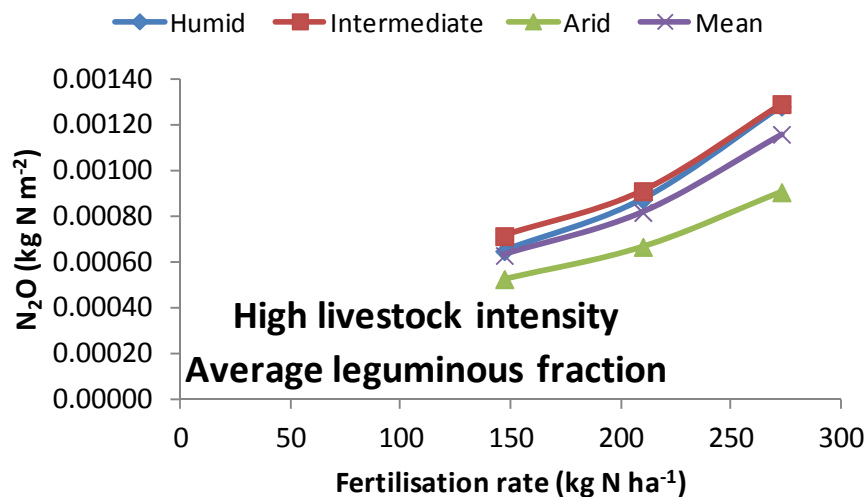
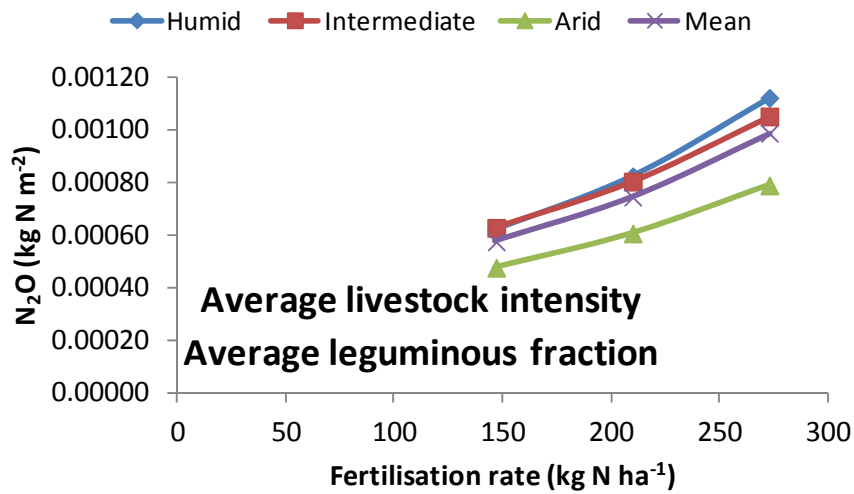
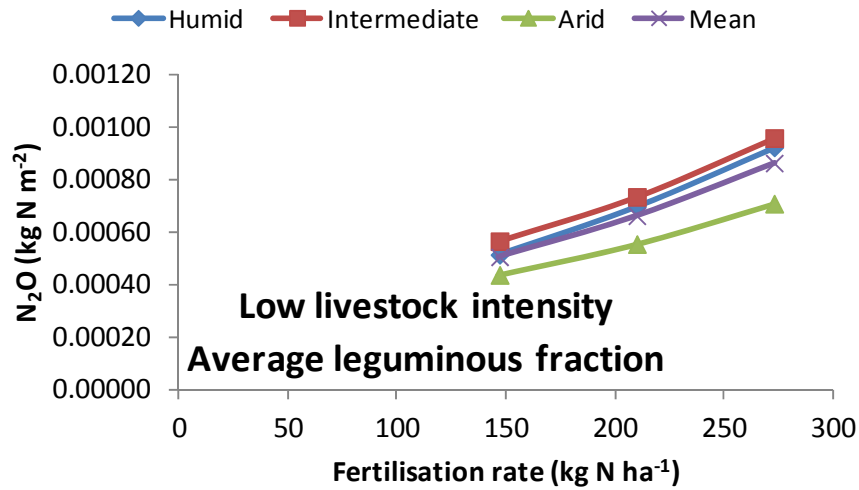


Figure 5.8. Annual nitrous oxide ( $N_2O$ ) emissions simulated by PaSim at Laqueuille (France) for contrasting years (from arid to humid) and for alternative nitrogen fertilisation rates and grazing animal intensities.

## 5.5. Conclusions

This study confirms that a relatively simple approach to mitigate both anthropogenic N<sub>2</sub>O and CH<sub>4</sub> emissions in managed grasslands systems is to reduce animal livestock, and hence enteric CH<sub>4</sub> emissions and the amount of excreta (Saggar *et al.*, 2008). Best management practices for mitigation of N<sub>2</sub>O emissions also include improvement of overall N management practices. However, the study also confirms the difficulty to sequester C in grassland soils by changing management practices, also considering that projected increasing frequency of drought and heat wave events may turn grasslands into C sources, contributing to positive carbon-climate feedbacks (Ciais *et al.*, 2005; Soussana *et al.*, 2007).

The results obtained from this simulation study depend on the site and climatic conditions (aridity, precipitation, temperature, altitude). General conclusions are that:

- 1) decreasing N fertilisation rates and density of grazing animals can be envisaged as options to reduce emissions of N<sub>2</sub>O and CH<sub>4</sub>, respectively;
- 2) CO<sub>2</sub> emissions (NEE) are highly affected by the variability of climate conditions, indicating that grassland sites may become sources of C (or may reduce their sinking rates) in arid years, regardless of the management;
- 3) increasing/decreasing clover fraction in the sward is ineffective in terms of GHG emissions, but it can become a valuable option in the perspective of reducing inputs from N fertilisers.



## 6. From animal to field level with process-oriented modelling: case studies for dairy farming

The effect and consequences of measures taken to mitigate GHG emissions often extend the boundary of a single component or level of aggregation of the farming system (*i.e.* crop, field/paddock, animal/herd, manure, housing). For this reason, it is important to investigate the effect of such mitigation measures in sufficient detail. Attempting to make an inventory of the effect of measures that can be taken on a specific farm hence requires a detailed analysis. Default emission factors have been developed for national surveys of GHG emissions and for accounting of national GHG budgets (Tier 1 and Tier 2 approaches; IPCC, 2006). They serve to indicate an average effect for an average farm, but they have not been developed for detailed and case-specific evaluations of measures, nor do they account for variation among farms and conditions (Bannink *et al.*, 2014). In this respect, process-oriented models have more promise as they represent the underlying mechanisms that drive GHG emissions and are less bound by the empirical datasets they have been derived from.

A combined use of such detailed, process-oriented models for different components or levels of aggregation demands a feasible exchange of model inputs and outputs. The methodology for such combined use of models for studying GHG emission related to enteric fermentation, manure storage and soil processes is described in paragraph 6.1. Models used are the Dutch Tier 3 for enteric fermentation, a manure storage model and DNDC. In the subsequent paragraph 6.2 a combined use is demonstrated for four contrasting dairy farms cases which were frequently and intensively monitored and for which reliable data about farming practice and performance were available. The cases selected vary widely in intensity of farming (number of cows ha<sup>-1</sup>), intensity of N fertilisation (kg N ha<sup>-1</sup> yr<sup>-1</sup>) and intensity of feeding (DM intake and milk yield, kg cow<sup>-1</sup> yr<sup>-1</sup>).

### 6.1. Format for combined use of process-oriented models

The Dutch Tier 3 for prediction of enteric CH<sub>4</sub> in cows was extended to represent details of C and N fractions excreted in urine and faeces (urine fractions distinguished from faecal fractions), the various C and N fractions that can be identified in excreta, sulphur excretion, and urine and faeces volumes. Based on earlier work by Reijs (2007) and Ellis *et al.* (2011) equations were added to the model to quantify quantity and composition of excreta and these model outputs were made compatible with the inputs required by the newly developed model of manure storage (Hutchings *et al.*, unpublished). By this extension of the Dutch Tier 3 model (Bannink *et al.*, 2011) all (manure-related) inputs that are required to run the manure storage and the soil model could be generated. Table 6.1 summarizes the details on predicted manure production with the Dutch Tier 3 and the inputs required by the manure model (Hutchings *et al.*, unpublished) and the DNDC soil model (Li *et al.*, 2011).

Inputs for the enteric fermentation model have been derived from estimates of dietary DM intake, diet composition, and feed analyses. Rumen in situ degradation characteristics for the starch, protein and fibre component of dietary DM have been estimated based on reported digestibility estimates available for forages, and standard values for the ingredients in concentrates. For all farm cases such data were available for the main components in the diet. These estimates are sensitive to variation in growing conditions, fertilisation, forage harvesting, and conservation management. Next to the model outcomes of CH<sub>4</sub> emission and milk yield, output is generated for excreta volume and composition. This output includes the amount as well as the various nitrogenous and organic matter (carbonaceous) fractions excreted with urine and faeces. Several nitrogenous components in urine (urea) and faeces (feed crude protein, microbial, endogenous) were identified.

Table 6.1. Inputs required and outputs generated by models for enteric CH<sub>4</sub> emission (Dutch Tier 3), emissions from stored manure (manure model), and emissions from soils including carbon sequestration (DNDC).

<b><i>GHG source</i></b>	<b><i>Inputs</i></b>	<b><i>Outputs</i></b>
<b><i>Enteric</i></b>	<p><i>DM intake</i></p> <p><i>Chemical composition</i></p> <p><i>Intrinsic rumen degradation</i></p> <p><i>S intake, salt intake (Na, K)</i></p> <p><i>Milk composition</i></p>	<p><i>Milk yield</i></p> <p><i>CH<sub>4</sub> emission</i></p> <p><i>Feecal organic matter</i></p> <p><i>Fibre (dietary)</i></p> <p><i>Microbial</i></p> <p><i>Endogenous</i></p> <p><i>Nitrogen &amp; carbon fractions (incl. lipid) in urine &amp; faeces</i></p> <p><i>Ash and S excreted</i></p> <p><i>Urine &amp; faecal volume</i></p>
<b><i>Manure</i></b>	<p><i>Lignin, C, N and S</i></p> <p><i>Inert, slow &amp; fast degradable C and N fractions</i></p> <p><i>Total ammoniacal N</i></p> <p><i>Manure volume</i></p> <p><i>Manure residence time</i></p>	<p><i>NH<sub>3</sub>, N<sub>2</sub>, N<sub>2</sub>O, H<sub>2</sub>S &amp; CO<sub>2</sub> &amp; CH<sub>4</sub> emissions</i></p> <p><i>Conversion organic C &amp; N</i></p> <p><i>Organic matter digestion</i></p>
<b><i>Soil</i></b>	<p><i>C and N application</i></p> <p><i>Crop residues</i></p> <p><i>Meteorological data: temperature, precipitation, wind speed, RH, solar radiation</i></p> <p><i>Soil characteristics: Soil C content, nitrate and ammonium, clay content, bulk density, soil C pools, microbial activity, slope</i></p>	<p><i>Leaching NH<sub>4</sub><sup>+</sup>, NO<sub>3</sub><sup>-</sup></i></p> <p><i>NH<sub>3</sub>, CH<sub>4</sub>, CO<sub>2</sub>, NO, N<sub>2</sub>O, N<sub>2</sub></i></p> <p><i>Soil organic C and N sequestration</i></p> <p><i>Ecosystem water balance</i></p>



## 6.2. Simulation case studies (Dutch Tier 3 enteric CH<sub>4</sub> & DNDC)

The interdependence of the effect of measures on different sources of GHG emissions or on GHG emissions from the various farm components (animals, manure, soils) was studied by exchange of inputs and outputs of process-oriented models for three specific cases of dairy farms. Criteria for selecting these farm cases were the quality of the monitoring data that were available, the fact that one of them (the De Marke farm) also served as a show-case farm for AnimalChange and has been studied using the FarmAC model (WP9 and WP10 of AnimalChange; allowing comparison of modelling results and FarmAC results), and the contrast in farm management and farming conditions. High quality and resolution in time of monitoring of farm management (activity data) is crucial for a reliable setting of conditions and inputs for simulations with the process-oriented models.

The three cases studied are profiled in Table 6.2 and include two intensive dairy farming systems and one extensive system of farm management. These three cases are highly contrasting in applied management with respect to grazing, imports of artificial fertiliser, and animal stocking and milk production density per hectare.

### **Farm case 1; no grazing (total confinement), high fertilisation, intensive dairy farming.**

This case involves a well-monitored, intensively managed Dutch dairy farm (project Cows & Opportunities, 2012; <http://www.wageningenur.nl/en/project/Cows-and-opportunities>) in the South West of the 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.

### **Farm case 2; restricted grazing (mainly confinement), low fertilisation/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) (Cows & Opportunities, 2013). Partial grazing is applied during the Summer period for 133 d yr<sup>-1</sup> on average for 6 h d<sup>-1</sup>. A mixed farming practice is employed, characteristic for this region and this type of soil, which results in a mixed diet of grass herbage, grass silage, maize silage, ensiled corn cobs and purchased concentrates. The farm has a long history as a research station focussing on demonstrating how to improve dairy farming practices while largely eliminating artificial fertiliser inputs and hence minimising emissions to the environment (nitrogen and phosphorous, and GHG emissions), while keeping milk yields that can still be considered as representative for the farming practices in this region.

### **Farm case 3; unrestricted grazing, low vs. high fertilisation, extensive dairy farming.**

This case involves the Teagasc / Tipperary Co-op joint demonstration research farm in Solohead of Teagasc. Tipperary is a working farm where different management options with respect to grassland, grazing and herd management are tested (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. Two different grassland management systems were adopted; a relatively intensive management with a high rate of N fertilisation and a grass sward mainly composed of perennial ryegrass, and a less intensively management with a low rate of N fertilisation and a grass sward with a relatively high proportion of clover.

Table 6.2. General characteristics of three contrasting cases of dairy farm management

Characteristics	Case 1 Dutch No grazing High N fertilis.	Case 2 Dutch Partial grazing Low N fertilis.	Case 3 Irish Unrestricted grazing High/ low N fertilis.
Area (ha)	38	55	11
Area grassland (% total area)	70	61	100
Fertilization rate (kg N ha <sup>-1</sup> yr <sup>-1</sup> )	120	230	96 / 226 (22 / 6% clover)
Dietary grass : maize silage	1 : 3.5	1.4 : 1	1 : 0
DM intake (kg DM d <sup>-1</sup> )	19.9	18.5	14.6 / 14.3
Density (# lactating cows /ha)	3.4	1.5	2.2
Dietary concentrate (%DM)	32 (& 6 byprod)	21 (& 23% other)	10
Milk yield (kg/d per cow)	26.8	22.9	17.9

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

Model simulations have been performed with process-oriented models for enteric fermentation (Bannink *et al.*, 2011) and the DNDC model (Li *et al.*, 2011). Both models are described in detail in Chapter 2. Simulation results were obtained by ensuring that farm monitoring results (*i.e.* the activity data on animal numbers, milk yield, intake of harvested and purchased feed, crop yields, and manure production and application) were closely reproduced. Furthermore, a direct comparison of the simulated GHG emissions associated with milk production among farm cases required focusing on the lactating dairy herd only, excluding the effects of variation in GHG emissions that relate to variation in numbers of young stock per lactating cow and their nutritional management. The latter significantly contributes and may have profound impact on total on-farm GHG emission (or even GHG emission external of the dairy farm with offset of young stock to other farms) which partially needs to be allocated to milk production as well. However, it is argued here that variation in young stock management, in genetic merit and longevity of dairy cattle can be accounted for rather independently from the effect of variation in farm management, cow feeding and cow performance, and the GHG emissions associated with this. For this reason, the choice was made to restrict the combined application of the process-oriented models to variation in farm management related to milk production specifically.

Simulations of enteric fermentation were performed for the annual average of the diet, feed intake and cow performance. Although results for such an annual average may differ from a weighted average of simulations for seasons separately, earlier simulations show that the results are to a very high extent additive (Bannink, 2011), and hence bias in simulated enteric CH<sub>4</sub> as a result of considering annual instead of seasonal rations is relatively small. Furthermore, a large part of the diet is composed of silages from large silos which serve as a feed source throughout the year. Simulations of soil emissions and C sequestration were kept dependent on seasonality of conditions (*e.g.* temperature, precipitation, humidity) and farm management (*i.e.* fertiliser and manure application, harvesting, grazing). The outputs from the process-based model of enteric fermentation, the soil and the manure storage model could be linked as the C and N excretion amounts/rates generated from the enteric fermentation model could be directly inputted into the soil model in terms of a) pasture, paddock and range fertilisation and excreta/manure application rates, and b) amounts and C/N ratios of applied organic manure.

### 6.2.1. The case of no grazing (total confinement), intensive dairy farming

#### **Farm description**

This first farm case is characterised by a very intensive management which results in diets mainly composed of maize silages and concentrates. The farm does not apply grazing. Table 6.3 summarizes the diet of the herd on this farm which includes lactating cows as well as young stock during the monitoring period of the year 2012.

*Table 6.3. Annual average of herd characteristics, dietary characteristics, fertilisation management, soil characteristics and climatic conditions for the simulation case of a no grazing, intensive dairy farm (Cows & Opportunities, 2012; intensively monitored commercial farm in the South West of the Netherlands).*

<i>Herd</i>	<i>Soil, fertilisation, climate conditions</i>	
Lactating cows 132	Well-drained, dry, light sandy soil	
Young stock 56	27.4 grassland (13.5 t DM ha <sup>-1</sup> ; 374/193 kg N org/inorg)	
1291 t milk	13.0 arable land (maize) (18.7 t DM ha <sup>-1</sup> ; 168/25 kg N org/inorg)	
4.0% milk fat		
3.4% milk protein	Moist temperate climate	
<i>Diet lactating cows</i>	<i>Dietary composition (% DM) lactating cows</i>	
DM intake 20 kg DM d <sup>-1</sup>	Crude protein	16
Milk yield 27 kg d <sup>-1</sup>	Starch	26
0% grass herbage	NDF	39
21% grass silage	Crude fat	4
44% maize silage	Organic matter	94
28% concentrates		
7% by-products		

#### **Enteric methane**

The highly intensive feeding management resulted in a diet with a relatively low crude protein and fibre content and a high starch content. Consequently, cows captured ingested nitrogen in milk with an efficiency of over 30% on annual basis (hence including dry period). Such a high content of maize starch which is relatively resistant to rumen fermentation (bypassing the rumen) while being well digested in the intestine, contributes to a relatively low CH<sub>4</sub> emission per unit of ingested feed (Mills *et al.*, 2001; Bannink *et al.*, 2006) which is on average around 20 g CH<sub>4</sub> kg DM<sup>-1</sup> intake (Bannink *et al.*, 2011).

The monitoring data did not allow for a distinction in the feed allocated to the lactating cow herd and to young stock. Therefore, three extreme scenarios were simulated to correct for allocation of feed to young stock and to investigate to what extent simulation results for the lactating herd depend on assumptions made for feeding of young stock. The three extremes tested were:

- 1) young stock receiving a diet with the same composition as that for the lactating herd,
- 2) young stock receiving a diet which resembles the annual average for the Netherlands (Tamminga *et al.*, 2004) with silages assumed to be fed in proportion to the proportion of silages fed annually on that whole farm,
- 3) young stock receiving a diet which resembles the annual average for the Netherlands (Tamminga *et al.*, 2004) but the silages of lowest quality on the farm were being fed to young stock.

The simulation results for enteric CH<sub>4</sub> emission by the lactating dairy herd are given in Figure 6.1 and show that assumptions on young stock feeding had a very small influence on simulated emission for the lactating herd. The results in CH<sub>4</sub> per kg DM intake or kg milk differed less than 1%. For the most feasible scenario 3, where the highest quality silages are fed to lactating cows, the CH<sub>4</sub> emission was 12.0 g kg<sup>-1</sup> milk and 18.8 g kg<sup>-1</sup> DM intake. The value per kg milk is 33% lower than the 15.1 g CH<sub>4</sub> kg<sup>-1</sup> milk simulated with the same process-oriented model for the average Dutch dairy farm in 2012, but it is actually 4% higher than the national average of 18.1 g CH<sub>4</sub> kg<sup>-1</sup> DM intake (Bannink *et al.*, unpublished). Simulated values with the Tier 3 approach are 13% lower compared to prediction with the Tier 2 approach which adopts 6.5% of gross energy intake is emitted as CH<sub>4</sub>. These results reflect the quality of the diet, its high digestibility and the high milk yield per cow achieved within this farming management. The process-oriented model predicted for this farm that 5.6% of gross energy intake becomes emitted as CH<sub>4</sub> which is 14% lower than adopted with the Tier 2 approach.

The simulation results demonstrate the need to abandon a Tier 2 approach when the aim is to address and compare GHG emission in specific farm cases. Tier 2 approaches have been developed with the aim to perform for national inventories on GHG emissions, but not with the aim to be applied in studies where case-specificity is mandatory.

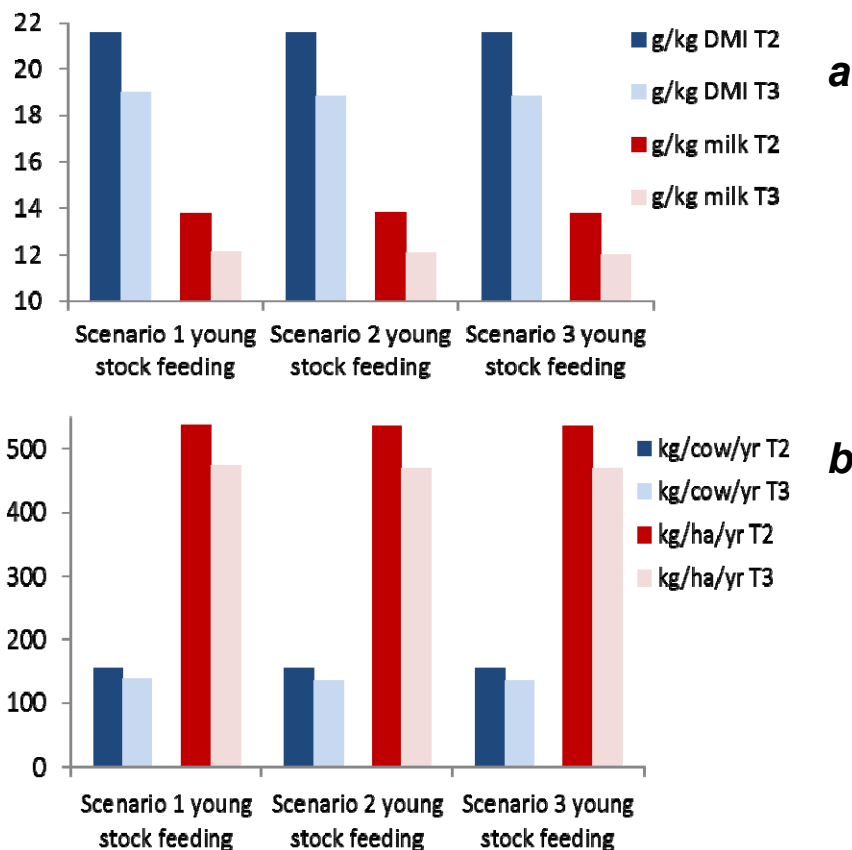


Figure 6.1. Simulated effect of assumptions of young stock feeding on the simulated enteric CH<sub>4</sub> emission in the lactating dairy herd (a) per kg DM intake or per kg milk, or (b) per cow or per hectare, for an intensively monitored commercial dairy farm in the year 2012 with high fertilisation rates, no grazing and intensive feeding management. Effects were calculated with the Tier 2 approach (T2) and a process-oriented model which is used as the Dutch Tier 3 for CH<sub>4</sub> emission in dairy cows (T3).

### Soil N<sub>2</sub>O emissions and C sequestration

Soil direct and indirect N<sub>2</sub>O emissions were calculated using IPCC default emission factors and outputs of the DNDC model. Indirect N<sub>2</sub>O emissions were derived from leached N and ammonia volatilisation, assuming that all lost N was locally re-deposited and applying the emission factor generated for direct N<sub>2</sub>O field emissions. There was a significant discrepancy between default and modelled direct and indirect N<sub>2</sub>O emissions, with soil N<sub>2</sub>O emissions ranging from 6.8 kg N<sub>2</sub>O-N ha<sup>-1</sup> yr<sup>-1</sup> to 8.7 kg N<sub>2</sub>O-N ha<sup>-1</sup> yr<sup>-1</sup> for Tier 2 and 3 simulations respectively (Figure 6.2). The differences mainly arose from direct N<sub>2</sub>O emission estimates generated from both Tiers. Higher DNDC modelled estimates arose due to a high soil denitrification rate as a consequence of soil texture and soil organic carbon content. Higher clay content soils will reduce hydro conductivity and preferential flow for N leaching and increase rates of denitrification to both N<sub>2</sub>O and N<sub>2</sub> (Clough *et al.*, 2003).

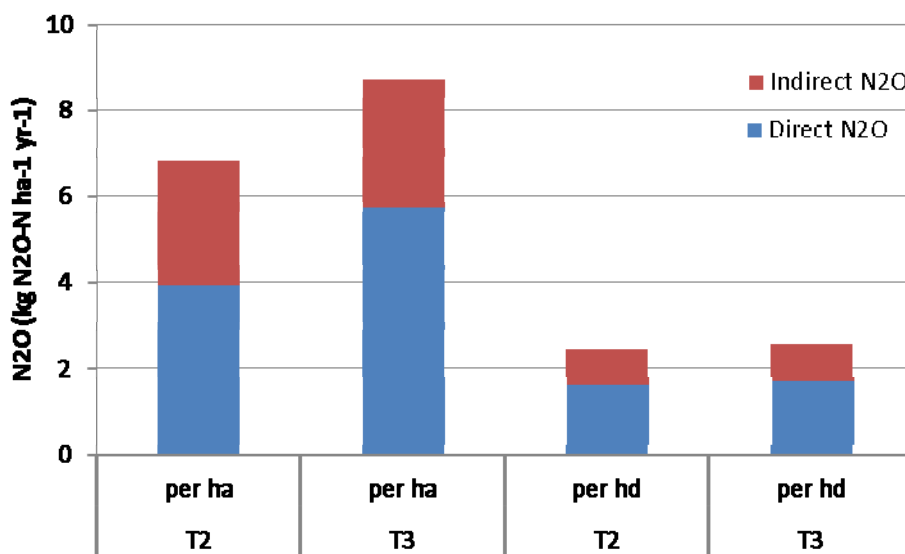


Figure 6.2. Field N<sub>2</sub>O emissions expressed per unit area and per head basis for high input no grazed (confinement systems) generated using Tier 2 emission factors and Tier 3 (DNDC) modelling. Blue columns indicate direct N<sub>2</sub>O emissions and red columns indirect N<sub>2</sub>O emissions associated with ammonia volatilisation and leaching.

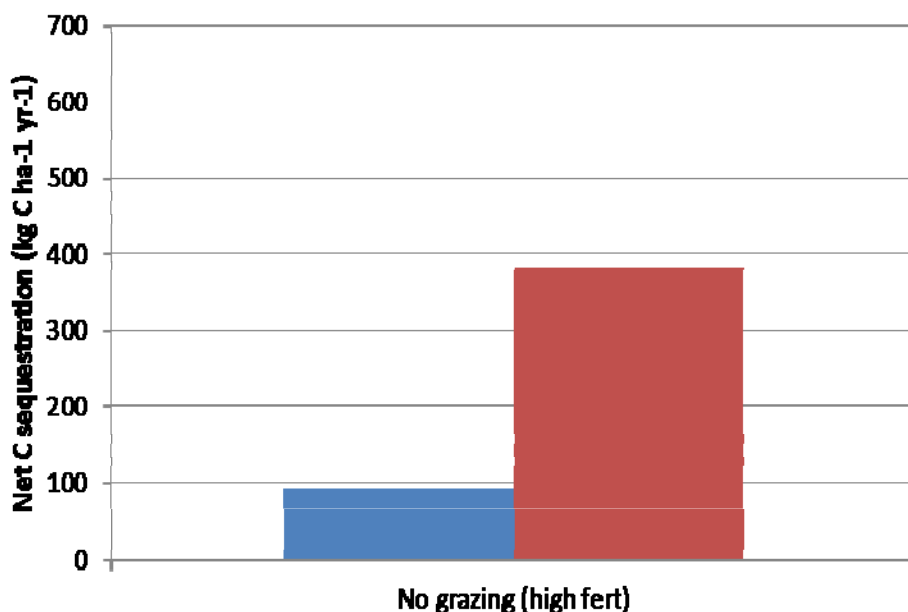


Figure 6.3. Net soil carbon sequestration associated with high input no grazed (confinement systems) Tier 2 emission factors (blue) and Tier 3 (red) modelling.

Tier 2 does account for soil type (either high or low activity clay, loam-based or sand-based). However, sequestration rates estimated under Tier 2 land-use factors were lower than measured values (Figure 6.3). This was due to the fact that soil organic carbon (SOC) under tillage and maize production, in particular, results in an annual SOC loss of  $1.1 \text{ t C ha}^{-1} \text{ yr}^{-1}$  (Ogle *et al.*, 2003). In addition, grassland had only been moderately improved, or improved with organic amendment. There is no impact of different levels of C input from manures in the Tier 2 approach, and neither does the Tier 2 account for the influence of climate, which in the Tier 3 DNDC modelling was observed to have a large impact on both arable and grassland C sink/source activity, to the extent that grasslands could flip between sinks and sources from year to year.

### 6.2.2. The case of restricted grazing (mainly confinement), low emission, intensive dairy farming

#### **Farm description**

This second farm case, experimental farm De Marke of Wageningen UR, is also characterised by an intensive nutritional management of the lactating cow herd, but it differs from the first farm case described in 6.2.1. in that farm management is optimized towards a minimal input of artificial fertiliser and a stocking density which is less than half that of the first farm case and partial grazing ( $133 \text{ d yr}^{-1}$  at  $6 \text{ h d}^{-1}$ ) is applied. About a quarter of the whole farm area is used for the production of maize and corn cob silage and partial grazing. This type of management also leads to a diet with a high proportion of maize products (maize silage and ensiled corn cob), as with the first farm case described in 6.2.1., but with a higher

proportion of grass products (grass silage and grass herbage). Table 6.4 summarizes the diet of the herd on this farm which excludes young stock in this case as the specific information on feed consumption by young stock was available, and this feed allowance was subtracted from total feed allowance to the whole herd to derive the feed allowance to the lactating dairy herd only.

### **Enteric methane**

This farm case is characterised by a very intensive feeding management which in combination with the low input of artificial fertiliser to that farm resulted in a diet with a low crude protein content, but also a relatively lower digestibility of the grass forage. As a result, cows captured ingested nitrogen in milk with an efficiency of over 30% on an annual basis which was equal to what cows achieved in the first farm case, despite the two percent units lower crude protein content of the diet. The starch content was slightly lower compared to that in the first farm case (6.2.1.).

The simulation results for enteric CH<sub>4</sub> emission by the lactating dairy herd are given in Figure 6.4 and show CH<sub>4</sub> emission was 14.1 g kg<sup>-1</sup> milk and 19.0 g kg<sup>-1</sup> DM intake. The value per kg milk is 7% lower than the 15.1 g CH<sub>4</sub> kg<sup>-1</sup> milk simulated for the average Dutch dairy farm in 2012 with the same process-oriented model as a Tier 3 approach, but it is actually 5% higher than the national average of 18.1 g CH<sub>4</sub> kg<sup>-1</sup> DM intake (Bannink *et al.*, unpublished). Simulated values were 12% lower compared to prediction with the Tier 2 approach which adopts 6.5% of gross energy intake is emitted as CH<sub>4</sub>. The process-oriented model predicted for this farm that 5.7% of gross energy intake becomes emitted as CH<sub>4</sub>.

These results are very similar to those obtained for the first farm case described in 6.2.1 despite the large differences in feeding management, in grassland management and forage production, and in intensity of dairy production per hectare.

*Table 6.4. Annual average of herd characteristics, dietary characteristics, fertilisation management, soil characteristics and climatic conditions for the simulation case of a restricted grazing, low emission, intensively feeding dairy farm (Cows & Opportunities, 2013; experimental farm De Marke).*

<i>Herd</i>	<i>Soil, fertilisation, climate conditions</i>	
Lactating cows 85	Well-drained, dry, light sandy soil	
Young stock 59	33.7 ha grassland (7.8 t DM ha; 294/51 kg N org/inorg)	
708 t milk	21.6 ha arable land; 15.3 maize (10.7 t DM ha <sup>-1</sup> ; 51 kg N org)	
4.5% milk fat	6.3 other	
3.5% milk protein	Temperate / continental climate	
<i>Diet lactating cows</i>	<i>Dietary composition (% DM) lactating cows</i>	
DM intake 18.5 kg DM/d	Crude protein	14
Milk yield 23 kg/d	Starch	23
8% grass herbage	NDF	44
25% grass silage	Crude fat	3
23% maize silage	Organic matter	93
21% concentrates		
23% other		



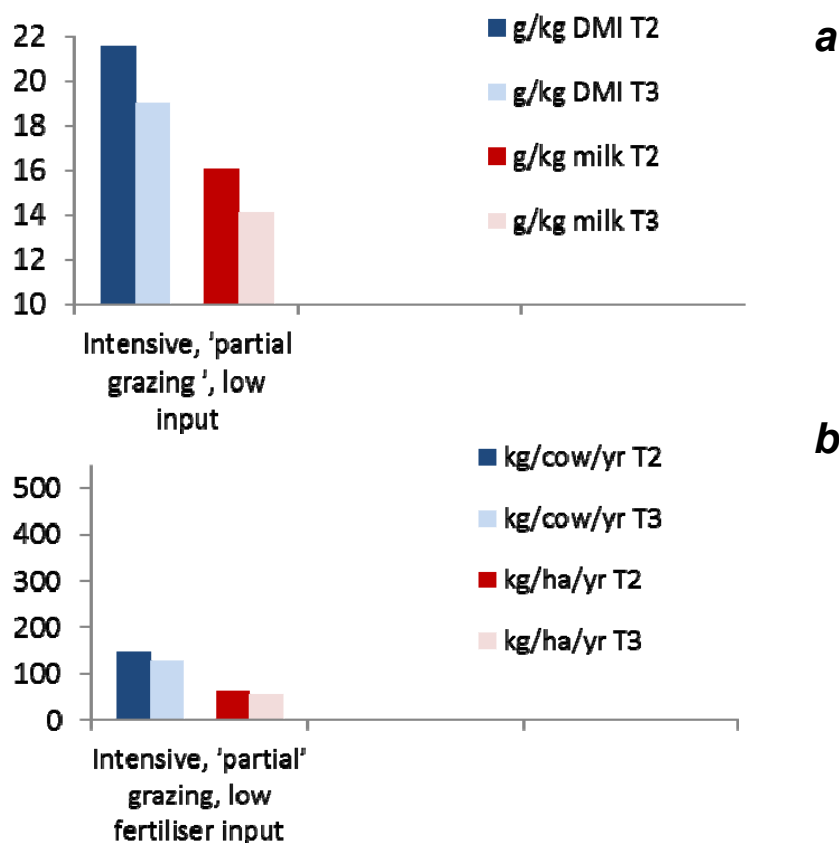


Figure 6.4. Simulated enteric CH<sub>4</sub> emission (a) per kg DM intake or per kg milk, or (b) per cow or per hectare on the De Marke farm in the year 2013 with a low artificial fertiliser input, restricted grazing (partial) and intensive feeding management. Effects were calculated with the Tier 2 approach (T2) and a process-oriented model which is used as the Dutch Tier 3 for CH<sub>4</sub> emission in dairy cows (T3).

### Soil N<sub>2</sub>O emissions and C sequestration

Soil direct and indirect N<sub>2</sub>O emissions simulated by either Tier 2 or 3 were observed to be relatively low compared to either the high input no-grazing systems (6.2.1) or the unrestricted grazing systems (discussed later in 6.2.3), when expressed on a per area basis (Figure 6.5). This was due to lower direct N<sub>2</sub>O loss and especially lower indirect N<sub>2</sub>O from leached N. However, emissions expressed on a per head basis were comparable with the high input system (circa 2 kg N per head), due to the higher stocking density. There was good agreement between Tier 2- and 3- calculated emissions. However, there were qualitative differences between lower and higher Tier simulations. Due to the fact that this farm was on a sandy free-draining soil the Tier 3-calculated N<sub>2</sub>O emission factor was under the 1% IPCC default factor. As a result of this, leached N was the major loss pathway simulated with DNDC and volatilisation was also a major loss pathway due to the fact that slurry application was a large proportion of applied N.

Carbon sequestration rates in the low input (mainly) confinement system were lowest compared to the other systems. Tier 2 land-use factors resulted in a lower sink compared to modelled results as in the intensive systems, due to the fact that land-use factors do not take into account a) the amount of organic amendment (C) applied to soil, b) the role of N in C sequestration and c) overestimate the soil organic carbon loss associated with maize production. The low C and N inputs compared with the high input system was the reason why the low input system had lower sequestration rates (Figure 6.6).



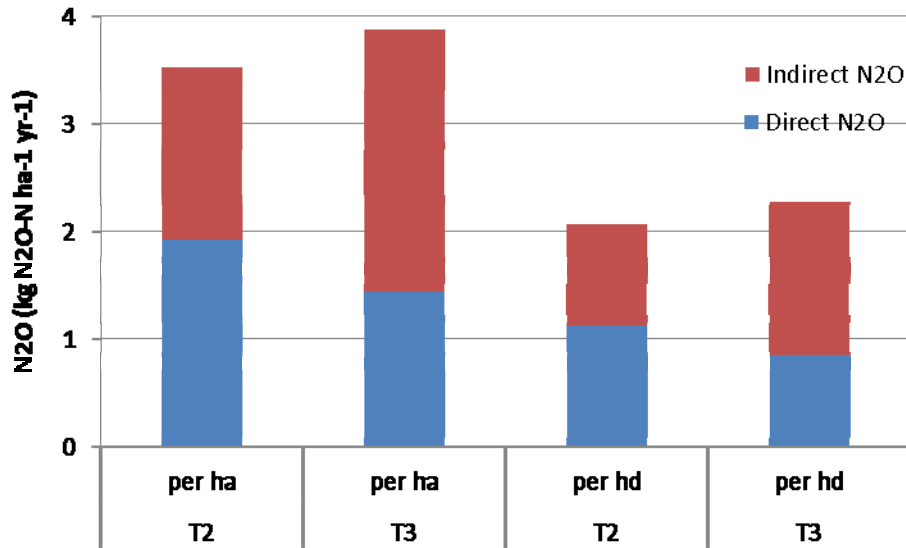


Figure 6.5. Field N<sub>2</sub>O emissions expressed on a per unit area and per head basis for low input no grazed, (mainly) confinement systems, generated using Tier 2 emission factors and Tier 3 (DNDC) modelling. Blue columns indicate direct N<sub>2</sub>O emissions and red columns indirect N<sub>2</sub>O emissions associated with ammonia volatilisation and leaching.

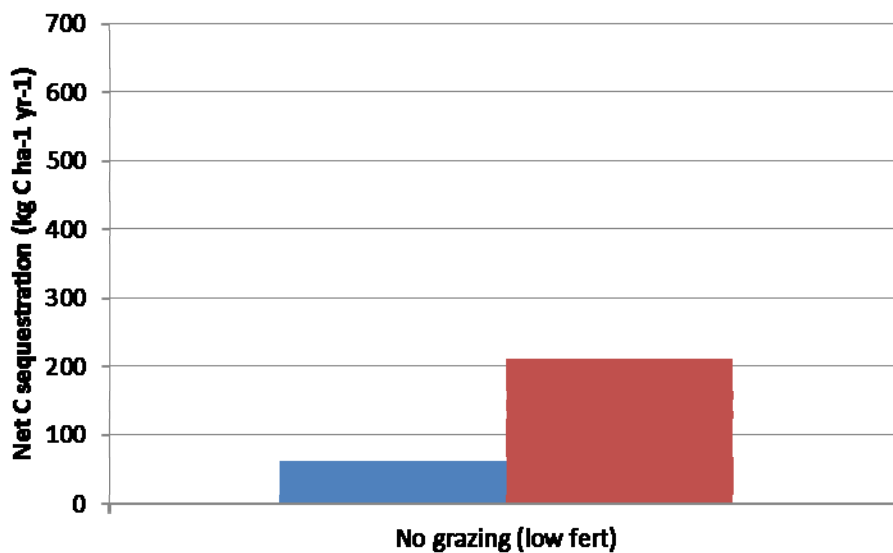


Figure 6.6. Net soil carbon sequestration associated with low input no grazed, (mainly) confinement systems, Tier 2 emission factors (blue) and Tier 3 (red) modelling.



### 6.2.3. The case of unrestricted grazing, extensive dairy farming

#### **Farm description**

The third farm case is Teagasc experimental farm Solohead that is characterised as an extensively grazed dairy farm. Half of the farm consists of *Lolium perenne* pastures, intensively fertilised (Table 6.5) whilst the other half consists of *Lolium/Trifolium* mixtures with low fertiliser application. Animals were grazed for 8 months of the year from early March to November at a 2.2 livestock units per hectare stocking rate. As a result, WSC and CP levels can vary considerably through the year. During housed periods, animal fodder principally consisted of grass silage, with some maize and concentrate. During the grazing period there was a small daily allowance of concentrates around milking, which remained less than 10% of daily feed intake (Humphreys *et al.*, 2009). Two different management options with respect to N fertilisation and grass sward composition were in place, which makes that this farm case can be split into two separate cases: one with strongly fertilised grassland (226 kg N ha<sup>-1</sup> yr<sup>-1</sup>) that is composed of perennial ryegrass, and one with a clover-rich (20% DM) grass sward and low N fertilisation rates (90 kg N ha<sup>-1</sup> yr<sup>-1</sup>) (Humphreys *et al.*, 2009). Only lactating cows were grazing on paddocks of 11 ha each.

*Table 6.5. Annual average of herd characteristics, dietary characteristics, fertilisation management, soil characteristics and climatic conditions for the simulation case of unrestricted grazing, intensively or extensively fertilised pasture on a dairy farm (experimental farm Solohead; Humphreys *et al.*, 2009).*

<i>Herd</i>	<i>Soil, fertilisation, climate conditions</i>
Lactating cows 24 ( <i>per system</i> )	Poorly drained Gelysol 11 ha <i>Lolium</i> grassland (13.4 tDM/ha; 34/192 kg N org/inorg), or 11 ha <i>Lolium/Trifolium</i> grassland (12.4 t DM ha <sup>-1</sup> ; 34/62 kg N org/inorg)
Young stock 0	
157 t milk ( <i>per system</i> )	Moist temperate climate
4.5% milk fat	
3.5% milk protein	
<i>Diet lactating cows</i>	<i>Dietary composition (% DM) lactating cows</i>
DM intake 13.8 kg DM/d	Crude protein 21
Milk yield 18 kg/d	
67% grass herbage	
23% grass silage	
10% concentrates	

#### **Enteric methane**

Predicted enteric CH<sub>4</sub> emission for grazing of a ryegrass or ryegrass/white clover pasture on this Irish farm was only 4% lower than estimates obtained with the Tier 2 approach that adopts a default 6.5% of gross energy intake being emitted as enteric CH<sub>4</sub> (Figure 6.7). Lactating cows utilized ingested N for milk protein production with 21% efficiency on annual basis. This is a lower percentage compared to the first and second farm case because of a

diet which consist almost exclusively of a highly digestible (OM digestibility around 80%, with a decline from 85% in spring to 75% in autumn) herbage with 21% crude protein content in DM, which is 5 to 7 % units higher than for the other farm cases.

Enteric CH<sub>4</sub> emission for these grazing systems was calculated to be 16.1 g kg<sup>-1</sup> milk and 20.8 g kg<sup>-1</sup> DM. These values are lower than the characteristic value of 20.4 g kg<sup>-1</sup> milk calculated for Irish dairy cattle with 110 kg CH<sub>4</sub> cow<sup>-1</sup> yr<sup>-1</sup> (calculated by the Tier 2 approach, adopting a standard 6.5% of gross energy intake being emitted as CH<sub>4</sub>) and 5400 kg milk per cow per year. The lower value on this farm is due to the 21% higher milk yield achieved with this particular farm case.

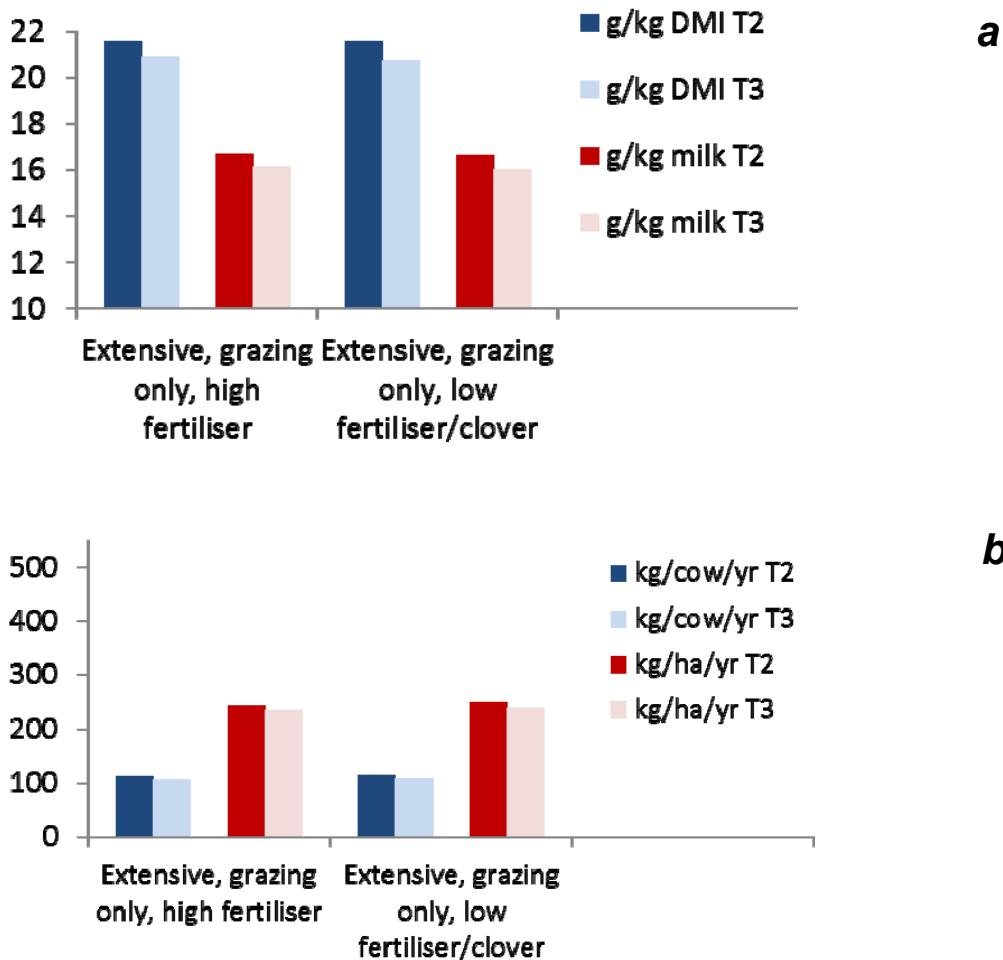


Figure 6.7. Simulated enteric CH<sub>4</sub> emission (a) per kg DM intake or per kg milk, or (b) per cow or per hectare on Solohead in the year 2013 with a low artificial fertiliser input, unrestricted grazing.

Effects were calculated with the Tier 2 approach (T2) and a process-oriented model which is used as the Dutch Tier 3 for CH<sub>4</sub> emission in dairy cows (T3).

### Soil N<sub>2</sub>O emissions and C sequestration

The Tier 2 and Tier 3 emissions are shown in Figure 6.8. Direct Tier 2 emission factors were 6.0 kg N<sub>2</sub>O-N ha<sup>-1</sup> yr<sup>-1</sup> and 4.7 kg N<sub>2</sub>O-N ha<sup>-1</sup> yr<sup>-1</sup> for *Lolium* only and *Lolium/Trifolium* mixtures respectively. This represented 21% reduction in emissions and reflected the fact that there was considerably less inorganic N input combined with the fact that the N<sub>2</sub>O emission factor for biologically fixed N is zero (IPCC 2006). By contrast, Tier 3 estimates of direct emissions were considerably higher, due to the fact that a) the soil type was an imperfectly drained gleysol and b) the year-round moist temperate climate in Ireland resulted in a water-filled pore space that rarely was less than 70%. This resulted in anoxic soils with high rates of partial denitrification. However, the extent of reduction between the high and low input systems was similar. Emissions based on per head basis and per unit milk basis were highest for the high-fertiliser grazed system, not only compared to the grass/clover system but also compared to the systems described in 6.2.1 and 6.2.2.

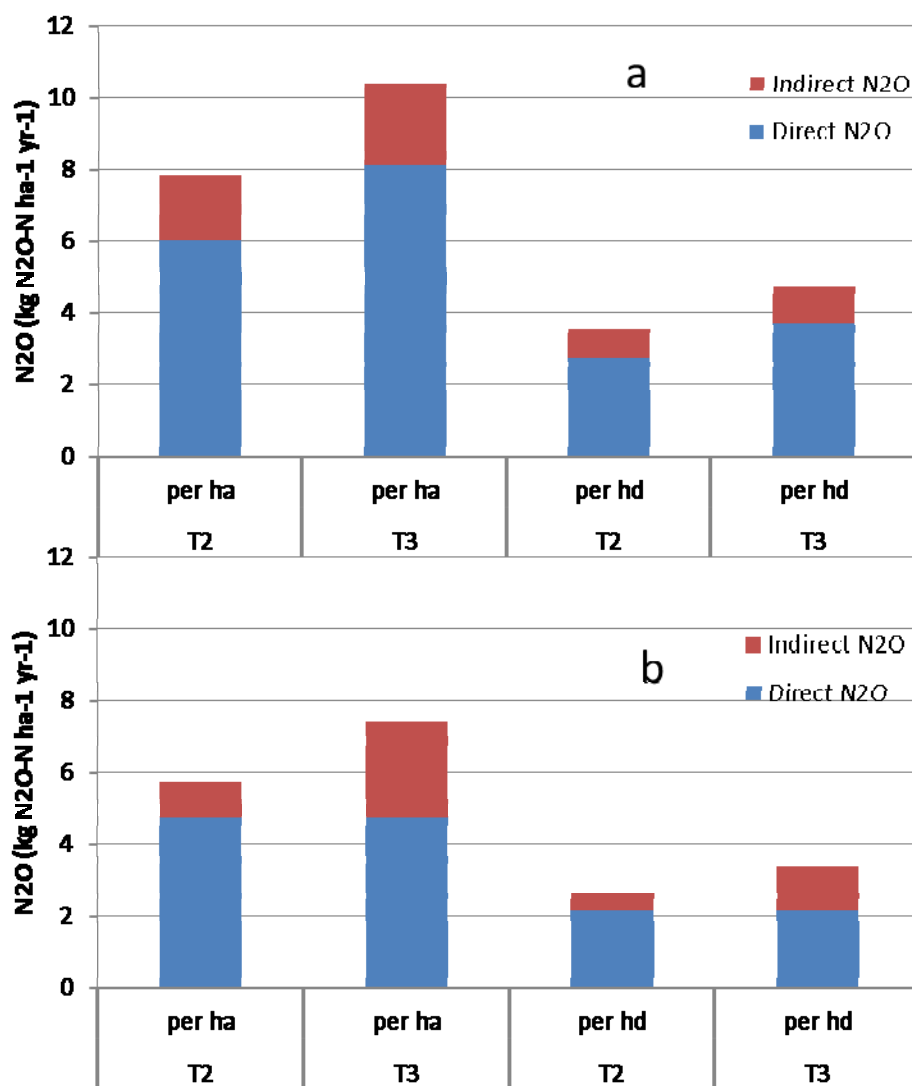


Figure 6.8. Field N<sub>2</sub>O emissions expressed on a per unit area and per head basis for a) high fertiliser input grazed pasture and b) grass/clover grazed pastures generated using Tier 2 emission factors and Tier 3 (DNDC) modelling. Blue columns indicate direct N<sub>2</sub>O emissions and red columns indirect N<sub>2</sub>O emissions associated with ammonia volatilisation and leaching.

Compared to the systems described in 6.2.1 and 6.2.2, Tier 2 land-use factors demonstrated much closer agreement with modelled SOC sequestration outputs particularly in the highly-fertilised grass only pasture (Figure 6.9). In contrast, grass-clover Tier 2 land-use factors overestimated SOC sequestration, principally as Tier 2 did not account for the relationship between C and N sequestration resulting from higher N inputs (Figure 6.9).

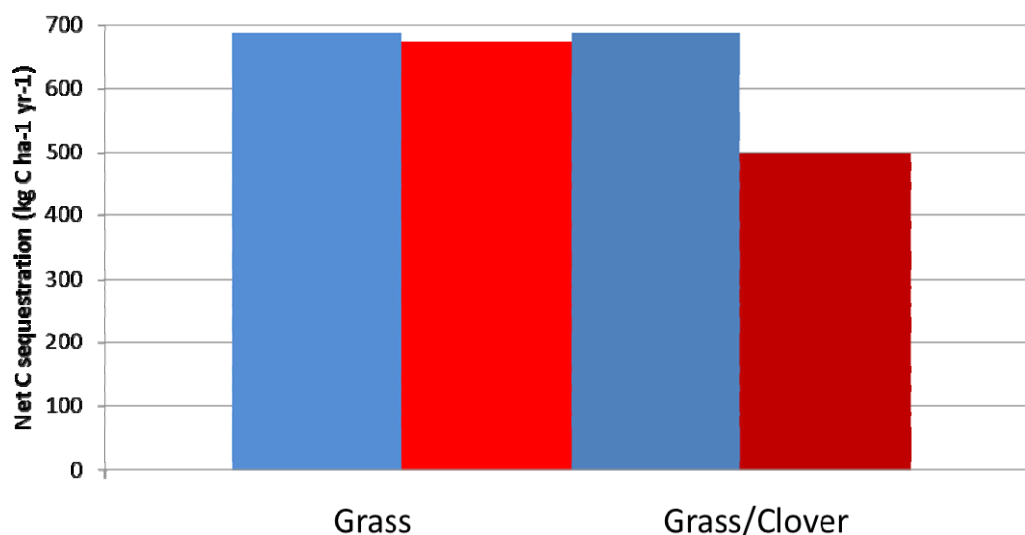


Figure 6.9. Net soil carbon sequestration associated with fertilised ryegrass only and grass/clover pastures using Tier 2 emission factors (blue) and Tier 3 (red) modelling.

#### 6.2.4. Comparison of GHG budgets of milk production in different farm cases

##### **Budget of on-farm GHG emissions**

An examination of emission intensities based on either Tier 2 or modelled Tier 3 outputs (Figure 6.10) indicate that the high input confinement system was the most C efficient, and this despite having little associated C sequestration across the whole farm due to maize cultivation. The low input system of 6.2.2 had proportionately less N<sub>2</sub>O per unit milk compared to high input system of 6.2.1 whilst the opposite held for CH<sub>4</sub>. The large difference between confinement/restricted grazing and unrestricted grazed systems was mainly driven by N application to soils, particularly pasture, paddock and range emissions (Figure 6.10). Clover addition reduced these losses and C sequestration offset emissions from grazed pastures by 20%. A lower Tier emission factors tended to overestimate emissions from confinement/restricted grazing systems but underestimate those from unrestricted grazed systems.

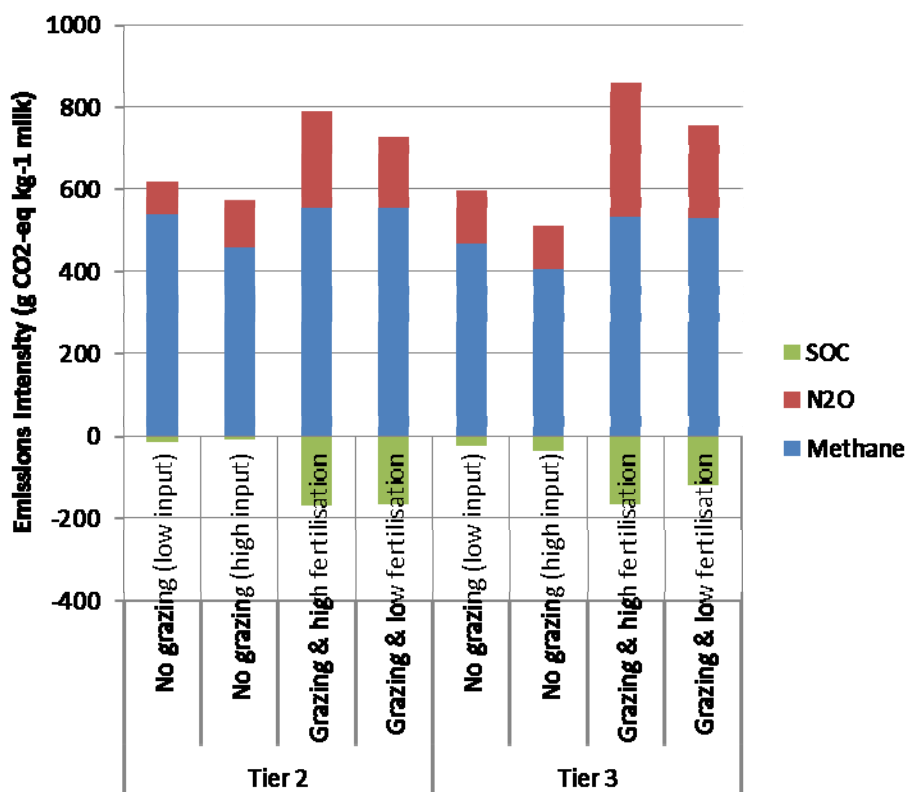


Figure 6.10. Simulated emissions  $\text{kg}^{-1}$  milk of  $\text{CH}_4$  and  $\text{N}_2\text{O}$ , and of  $\text{CO}_2$  from soil organic C (SOC), compared for partial grazing (mainly confinement) and low N fertilisation (farm case 2; indicated by “no grazing (low input)”), no grazing (total confinement) and high N fertilisation (farm case 1; indicated by “no grazing (high input)”), and unrestricted grazing with low or high N fertilisation (farm case 3; indicated by “grazing & low/high fertilisation”), calculated by a Tier 2 and a Tier 3 approach (indicated by “Tier 2” and “Tier3”, respectively). See 6.1 for further details on these farming systems.

## 6.2.5. Conclusions

With the selection of three farm cases (in total four management systems) a wide range in farming conditions was obtained with respect to 1) intensity of N fertilisation (animal manure, artificial fertiliser N), intensity of feeding dairy cows (unrestricted grazing, non- or restricted grazing; grass versus maize silage, concentrates), intensity of milk production (cow productivity), and farming intensity per hectare (stocking density; % grassland on farm; import of forages, concentrates, fertiliser). The process-oriented models predicted profound differences between farming systems for enteric  $\text{CH}_4$ ,  $\text{N}_2\text{O}$  emission from soils and C sequestration. Emission factors for  $\text{N}_2\text{O}$  and C sequestration predicted by process-oriented models were strongly different from IPCC Tier 2 defaults for all farm systems, whereas those for enteric  $\text{CH}_4$  were lower in particular for the more intensive feeding systems that included a higher proportion of maize silage and were less grass based than the extensive unrestricted grazing systems.

Carbon sequestration is an important part of the budget of on-farm GHG emissions. In particular for the grass-based farming systems it may compensate up to a quarter of the  $\text{N}_2\text{O}$  and enteric  $\text{CH}_4$  emissions. The  $\text{N}_2\text{O}$  emissions (direct and indirect) were particularly low for

the intensive systems including high proportions of maize silage, concentrates or other non-forage products in the diet of lactating cows. The difference in N<sub>2</sub>O emission between the non- or restricted grazing, intensive systems, and the unrestricted grazing, extensive systems was not totally compensated by the higher C sequestration with the latter.

Net GHG budget for the on-farm GHG emissions was lower for the intensive systems that did not or had far less grazing compared to the extensive unrestricted grazing systems. One of the reasons is the rather low crude protein content of the diet due to the relatively high proportion of maize and, in addition for the farm case that aimed for low N emission, the low rate of artificial N fertilisation. It has to be noted however that the net GHG budget does not include the GHG emissions that were associated to the maize silage and concentrates imported on the farm, and the higher use of fossil energy associated with the higher intensity under these farming conditions. Also not included in the net GHG budget is the possible trade-off of a higher CH<sub>4</sub> emission from manure with a higher proportion of maize silage in the diet, partly compensating the mitigating effect of maize silage on enteric CH<sub>4</sub>. Would all these effects be taken into account than the non-grazing, intensive farming systems may have similar or higher GHG emissions per kg of milk produced. Intensity of milk production per ha remains to be much higher on the intensive systems however.

There were clear and incidentally profound differences between emission factors according to IPCC Tier 2 methodology and predicted by the process-oriented models. This outcome suggests that adoption of Tier 2 methodology should be abandoned with surveys of the consequences of variation in farm management and farming conditions on GHG emissions, or for the farm-specific evaluation of the effect of mitigation options on GHG emissions. Adoption of Tier 2 methodology does not suit the latter objectives as it must be seen as instrumental at the scale for which it has been developed, which is surveys at a national scale and for average farming conditions.

The present study showed a combined use of process-oriented models which delivers a more detailed insight in individual on-farm GHG sources. This leads to the conclusion that these process-oriented models could well be consulted when assumptions need to be made on emission factors in models that describe GHG emission at a farm scale, or in Life Cycle Analysis of specific production chains or farming conditions.

### **6.3. Options at manure level; simulation case studies (manure model)**

#### **6.3.1. Mitigation options and manure emissions; sensitivity analysis**

The quantity and quality of excreta produced from the range of diets simulated with the Dutch Tier 3 for enteric CH<sub>4</sub> (Chapter 4) was used as input to the manure model. For the purpose of this exercise, the excreta were added once per day to the manure storage, with no addition of other material (spilt feed, bedding), spilt drinking water or washing water. Slurry was stored for 180 days with no crust formation and a manure temperature of 12 degrees Celsius was used. Methane emissions were simulated for different quality of grass-based diets (see 4.1 for further details) and the exchange of maize silage for grass silage (see 4.3 for further details) and for different DM intake levels. However, only results with a DM intake of 18 kg DM animal<sup>-1</sup> day<sup>-1</sup> are considered here. The annual simulated CH<sub>4</sub> emissions from the storage of manure generated by diets of grass silage receiving high or low amounts of N fertiliser or cut early or late in the season are shown in Figure 6.11.

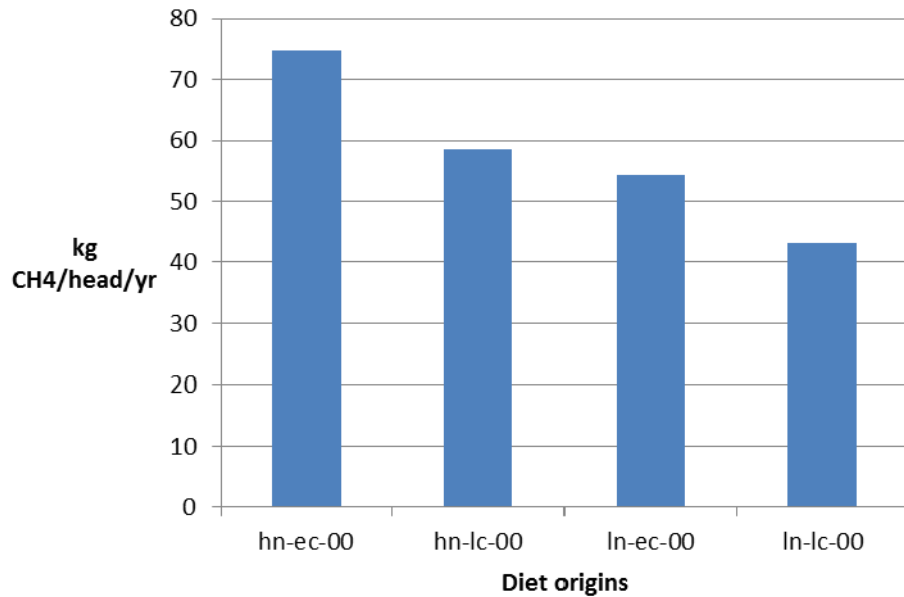


Figure 6.11. Variation in annual methane emissions from slurry with grass fertilisation and cutting time. Key: <grass fertilisation level> - <grass silage cut> - <maize silage cut>, hn=high nitrogen fertilisation, ln=low nitrogen fertilisation, ec=early cut, lc=late cut. See 4.1 for further details.

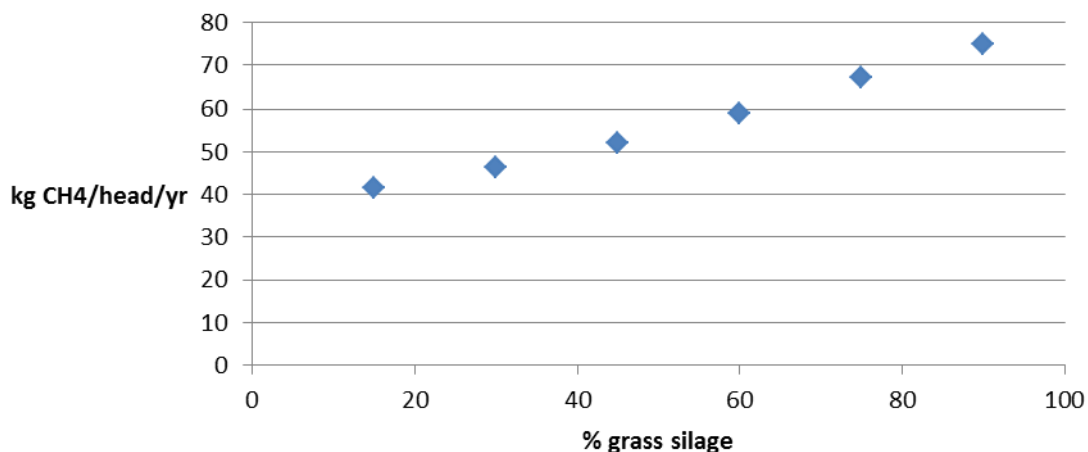


Figure 6.12. Annual methane emissions from slurry deriving from diets varying in proportions of grass silage. See 4.3 for further details.

The annual simulated CH<sub>4</sub> emissions from the storage of slurry generated by varying amounts of grass silage in the diet are shown in Figure 6.12. The results cannot be explained by changes in the amount of volatile solids excreted in association with the different diets (Figure 6.13). However, the results can be explained by relating the CH<sub>4</sub> emissions to the annual amount of rapidly-degraded organic matter excreted (Figure 6.14). Note that if the relationship were to be extended to 0% grass silage, there would still be an emission of CH<sub>4</sub>. This is because the rapidly-degraded organic matter is the main but not sole source of CH<sub>4</sub>; the slowly-degraded organic matter and VFAs also contribute. Because the comparison is made for diets in which the dry matter intake was identical, a decrease in the excretion of rapidly-degraded organic matter was associated with a large increase in the amount of slowly-degraded organic matter excreted and a small increase in the excretion of VFAs.



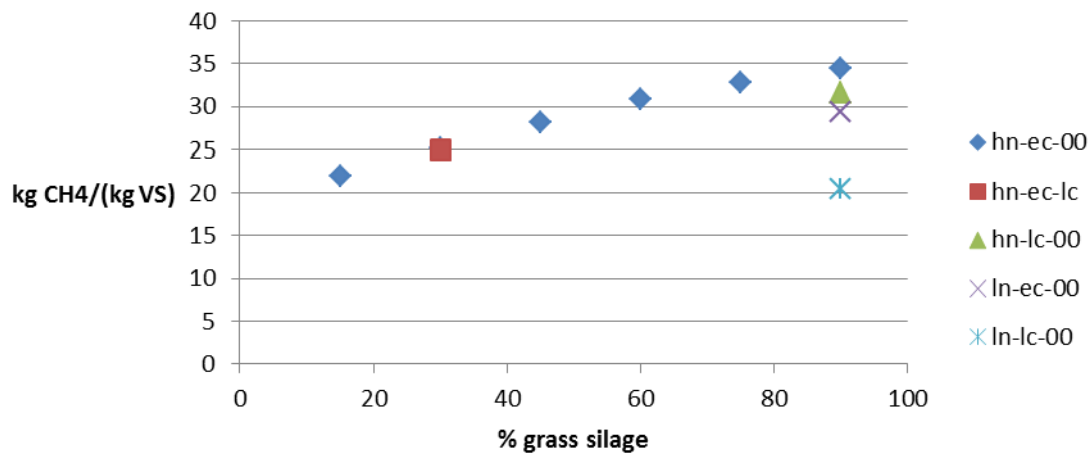


Figure 6.13. Methane emissions from slurry per unit volatile solids (VS) excretion for the different diets. See Figure 6.11 for key.

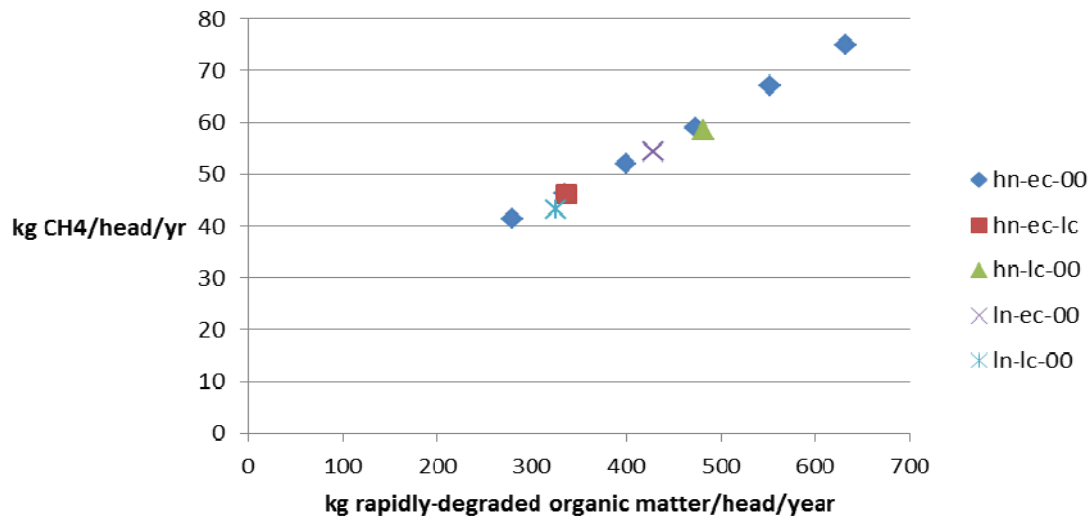


Figure 6.14. The relationship between the methane emission from the manure and the annual excretion of rapidly-degraded organic matter. For key see Figure 6.11.

The range for the Western Europe IPCC (2006) Tier 2 emission factors for CH<sub>4</sub> for dairy cattle manure in temperate conditions (34 to 75 kg cow<sup>-1</sup> yr<sup>-1</sup>) is of the same order of magnitude as predicted by the model. More work on model parameterisation is required, however, before firmer conclusions can be drawn on emission from manure and about emission factors to be applied for various farming conditions at animal, manure and field level.

### 6.3.2. Simulation of farm cases

The manure model was used for simulation of the four farm cases discussed in 6.2. The input of dry matter to the manure model consists of faeces, urine, feed waste (unconsumed feed that enters manure storage) and bedding. The water entering the manure storage includes water in the aforementioned dry matter plus washing water and spilt drinking water. Since data for feed waste, bedding, washing water and spilt drinking water were not available, these inputs were ignored here.

The manure model requires as input the mass of total ammoniacal N (TAN). The output from the animal model includes TAN but in practice, a large proportion of the urine N excreted will also be rapidly converted to TAN, either on the floor of the animal housing or in the manure storage. For this exercise, it was assumed that all urine N was converted to TAN in the animal housing. The TAN created on the animal house flooring will be subject to NH<sub>3</sub> volatilisation. To account for this volatilisation, an emission factor of 17% of TAN was assumed for all systems. Since the estimate of 17% is based on empirical measurements that will have included a contribution of NH<sub>3</sub> from manure stored in manure channels, the method includes an element of double accounting.

The input of data from the animal model consisted of the daily mass of excreta, dry matter, ash, ADL, NDF, lipid (assumed to equate to raw lipid) and crude protein. This information is used to partition the feed consumed into the pools used in the model; Fast, Slow, Inert, VFA and TAN. Specific C:N:H:O elemental ratios are assumed for each pool. The degradation of the pools is determined by temperature-dependent rate parameters and the duration of storage.

In this exercise, the input of manure from a nominal 100 livestock herd was assumed to occur over 180 days for farm case 1 with non-grazing and high N fertilisation, and for farm case 3 with unrestricted grazing and either low or high N fertilisation (see 6.2 for further details), and for 7 days at farm case 2 with restricted grazing (mainly confinement) and low N fertilisation (see 6.2 for further details) where manure is transported continuously to the anaerobic digester and manure volume in storage remains low. The composition of the manures from the farm cases, as used in the manure model, is shown in Figure 6.15.

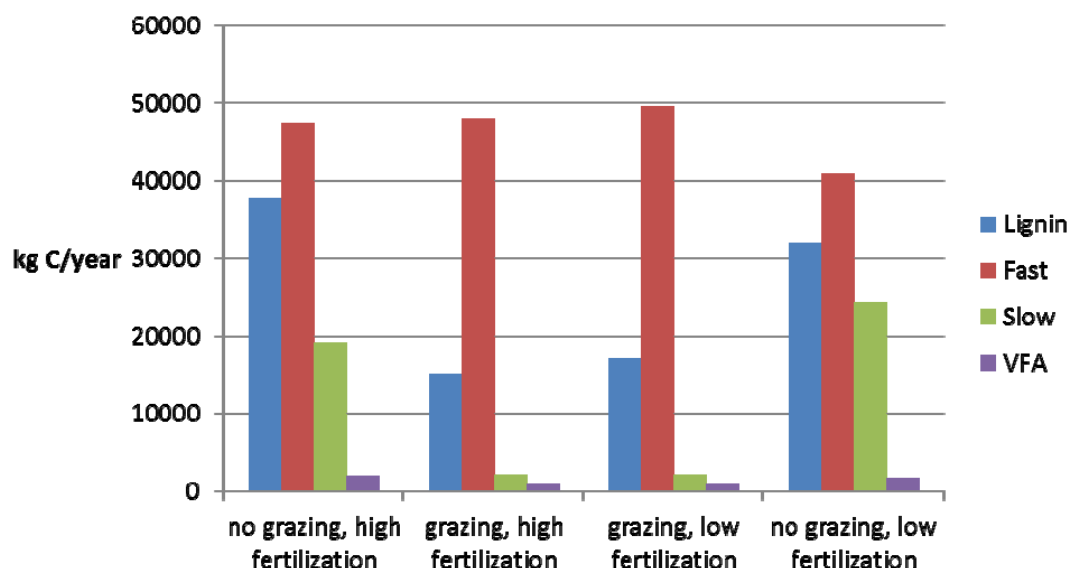


Figure 6.15. The composition of excreta entering the manure storage as simulated by the Dutch Tier 3 for enteric CH<sub>4</sub> used as an input for the manure model. See 6.2 for further details of the farming systems.

The manure from the intensively fertilized grassland of the Irish farming systems had lower mass and lower concentrations of lignin and slowly-degradable organic matter than with low fertilization rates. For the Dutch farming systems with intensive feeding and either low or high fertilisation, such an effect of N fertilization is obscured by the fact that the farm with high fertilization applies a far higher proportion of maize silage which contributed to manure lignin, whereas the farm with the low fertilisation applied a substantial proportion of corn cob silage which is low in lignin.

The CH<sub>4</sub> emissions predicted by the model for the different slurries are shown in Figure 6.16. The values calculated using IPCC (2006) Tier 2 are shown for comparison. The modelled CH<sub>4</sub> emission for the manure from farm case 2 with restricted grazing and low N fertilisation (see 6.2 for further details) was 1 kg cow<sup>-1</sup> yr<sup>-1</sup>, due to a very low residence time of the manure in storage prior to anaerobic digestion. The Tier 2 emission factor for this farm case would in practice not be appropriate for reporting but is shown in order to illustrate the consequences of this specific type of manure management. The IPCC (2006) Tier 2 manure emission factors and the apparent emission factors derived from the manure model are shown in Figure 6.17.

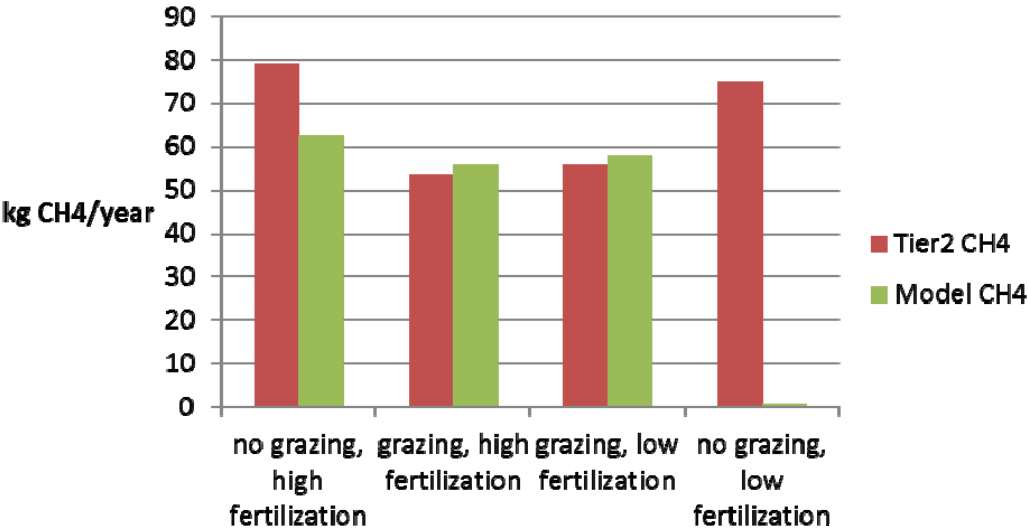


Figure 6.16. The CH<sub>4</sub> emission from manure according to IPCC (2006) Tier 2 and simulated by the manure model. See 6.2 for further details of the farming systems.

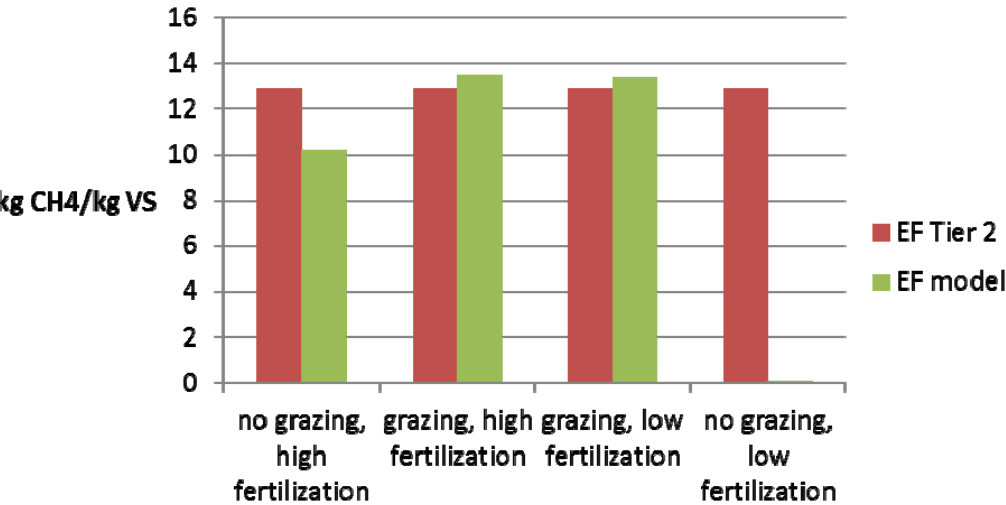


Figure 6.17. The manure CH<sub>4</sub> emission factor according to IPCC (2006) Tier 2 and simulated by the manure model. See 6.2 for further details of the farming systems.



### 6.3.3. Conclusions

The main conclusions of modelling manure emissions are:

- 1) The quality of the livestock feed affects the quality as well as the quantity of the excretion in a manner that cannot be adequately captured using the IPCC (2006) Tier 2 approach.
- 2) The variations in the quality of the excreta lead to differences in their capacity to generate CH<sub>4</sub> during storage.
- 3) The residence time of the manure has a strong impact on the emissions.
- 4) The simulations undertaken here did not include inputs of feed waste or bedding. Significant inputs of feed waste in particular could increase CH<sub>4</sub> emissions, since such feeds would normally contain a larger proportion of degradable organic matter than either excreta or bedding.
- 5) Note that these present simulations were undertaken with default parameters values for the manure model, that more work is required to validate these parameters and that the results have to be taken qualitatively.



## 7. Discussion

This chapter discusses the general findings for mitigation potential in different farming systems and with different metrics, and also within the perspective of adaptation to climate change for the case of extensive grassland-based systems. Outcomes obtained with different methodologies to estimate variation in prominent sources of on-farm GHG emissions will be discussed, including impact on estimated GHG farm budget.

### Intensity of livestock farming

There are main differences in intensity of ruminant production systems throughout Europe which have profound effects on GHG emissions from these systems. Differences in cattle productivity and intensity of milk or beef production per ha are achieved with the introduction of maize or other crops in the farming system and an increased purchase of concentrates and by-products, and by rate of manure and fertiliser application to soil, and by stocking density. Intensification pursued to increase feed intake which leads to a higher intake of metabolizable energy and a higher growth rate or milk yield per animal (Dijkstra *et al.*, 2013). These measures may strongly reduce enteric CH<sub>4</sub> emission and N emissions per kg of milk and increase efficiency of feed and N utilization as relatively less feed is required for maintaining the dairy herd and relatively more of every kg of feed consumed can be used for lactation or growth. Less animals are needed per unit of product, and hence less animals have to be kept for the same amount of animal product to be produced. However, whilst intensifying a farming system may inflate productivity per ha and inflate farm income, it also inflates costs of feed purchase and of maintaining the farming system. Furthermore, it inflates GHG emissions related to the loss of soil carbon by converting grasslands into arable lands for crop production (Vellinga *et al.*, 2011), and GHG emissions associated with the production and transport of purchased concentrates and by-products (off-farm emissions) and associated with maintaining a more intensive farming management which includes manure management. Because of the high intensity of farming, high levels of artificial fertiliser and cattle manure are applied, which intensifies emissions (including GHG) to the environment (Schils *et al.*, 2007).

Feasibility of measures and consequences and potential for intensification differs per region. In vulnerable regions or on certain soil types grass production may prove to be the only possible land use. Under more arid conditions the production and nutritive value of grass would prevent more intensive forms of livestock farming. Because dairy cattle require a more intensive feeding with higher nutritive quality of feeds, under more harsh and arid conditions other ruminant production systems than dairy production are normally more prevalent. Production of maize or other starch-rich crops might intensify dairy production under arid conditions and decrease some of the on-farm emissions, but this requires irrigation to overcome lack of soil water and raises the problem of loss of soil C in contrast to maintaining grasslands. Also under very humid conditions, an intensification of livestock production may lead to too high risks of excessive emissions to the environment through leaching.

Although intensification in itself may aid in reducing on-farm emissions per unit of animal product obtained, the impact of this on GHG emissions and the effect of mitigation measures that can be taken has always to be considered within the context of climatic conditions, the type of livestock production system under consideration and the off-farm GHG emissions required to maintain this system. Differences in sources of GHG emissions across farming systems may well be larger than the effect a mitigation measures can resort. Mitigation measures that can be taken involve 1) an altered cattle nutrition or allowance of forages or pasture with a different nutritive quality; 2) an altered manure storage and management; 3) an altered soil, water, pasture and cropping management; 4) an altered cattle herd

management. The present study identified the potential contribution of these measures to mitigate GHG emissions from enteric, manure and soil sources. Adaptation of farming systems may change the potential of these measures, but across systems or climate conditions the trend of mitigation effects sustains (for grass-based and non-grass-based; for extensive and intensive systems). Several options tested appeared fairly robust for a range of farming systems and sites, with exception of the more arid farming conditions. Nevertheless, the present investigation also demonstrates the variability of ruminant production systems in Europe and the impact of this on levels of CH<sub>4</sub>, N<sub>2</sub>O and CO<sub>2</sub> emission and C sequestration. These ruminant production systems tend to be far more variable than pig and poultry systems which use grain-based feed rations, and are more controlled and less variable. For ruminants, diets vary from the seasonally low-digestible extensively grown herbage of montane meat production systems to mainly maize silage and concentrates in intensive milk production systems.

### **Impact of systems and choice of metric on conclusions for different farming systems**

Comparisons of farm-gate GHG balances indicate that confinement systems had a lower GHG footprint compared to grazed systems when expressed on either an emissions intensity basis or a per unit livestock basis. However, this was highly dependent on the reporting Tier used. Without SOC included, high-input grazed systems had a 27% (Tier 2) and 40% (Tier 3) higher CH<sub>4</sub> and N<sub>2</sub>O emission compared with the high input (mainly) confinement system. These differences are consistent in trend but higher in terms of absolute differences with farm-gate life-cycle analyses (LCA) from other studies (Phetteplace *et al.*, 2001; Basset-Mensetal, 2009). However, once SOC sequestration was accounted, the differences reduced to 9% (Tier 2) and 31% (Tier 3). Indeed, it has previously been shown that if SOC sequestration is taken into account, the C footprint of livestock production can be reduced by 25% - 40% (Crosson *et al.*, 2011). It should be noted, however, that the differences between production systems are less clear once pre- and post-farm gate emissions are included in the systems analysis. Studies on full LCA of dairy systems have shown contradictory results with some showing a small advantage for confinement systems (Phetteplace *et al.*, 2001), whilst other demonstrate lower emissions intensity in extensive grazed systems (O'Brien *et al.*, 2010). Differences generally arise from a) the general assumptions made for both types of systems and b) whether or not specific example farms are being assessed. The range of values for individual farms across each production system has been shown to vary by >50% (Crosson *et al.*, 2011). Therefore, general assumptions can be unreliable and even more so when specific individual farm cases are studied. As a consequence, a large survey of individual farms is possibly the most robust approach to study GHG emissions. In light of this, we conducted the comparison between Tier 2 and Tier 3 (Tier 3-like) approaches to investigate the robustness of using different Tier emission factors to generate farm-scale emissions from both confinement and grazed systems. There was a large observed discrepancy between Tiers, due to the fact that the lower Tiers use ubiquitous emission factors. Differences between GHG emissions are driven by stock changes (*e.g.* differences in DM intake, N deposition onto soils, and C input), whilst the modelled outputs take local conditions (soil type and climate) into account. As a result, both the inputs and emission factors are altered. In the present study, the Irish grazed pastures had both heavier soils and a wetter climate resulting in much higher N<sub>2</sub>O emission factors compared to the default factors. Feedstock was less digestible also resulting in higher CH<sub>4</sub> losses. Nevertheless, the level of SOC lost under maize production was much higher for the Dutch farms when compared to the modelled outputs. Therefore, when comparing these specific farms, the Tier 3 modelled outputs were a more accurate reflection of the actual emissions from these farms and for deriving GHG balances. However, if making generalized comparisons, Tier 2 might be a truer reflection as there is no need to make explicit assumptions for each system on crucial parameter values which are unknown. Indeed, using Tier 2 resulted in little difference

in emissions intensity between farming systems. With the use of higher Tiers (process-oriented models) there is also a higher need of activity data (climate, soils, fertiliser application time, grazing dates), in particular for the soil emission models and to a lesser extent for the manure storage model (because of the less variable storage conditions and the isolation of manure from climatic influences), and the enteric fermentation model (the latter because of the active regulation of the enteric conditions by the ruminant). There is relatively less a problem for the modelling of enteric fermentation because of the (compared to soil and manure storage) well-controlled environment in the gastrointestinal tract of ruminants where fermentation processes take place and the fairly good documentation on feeds and forages, compared to the modelling of soil processes as these suffer from a high variability of management interventions, changing meteorological conditions, and seasonal changes.

Reductions in N inputs were reflected in a lower emissions intensity for the grazed systems, but not for the (mainly) confinement systems and this was independent of whether Tier 2 or modelled emission factors (Tier 3) were used. This is due to the fact that differences between the high and low input grazed systems were driven by N loss (especially pasture, paddock and range emissions), whilst differences between confinement systems were driven by feed characteristics and hence enteric CH<sub>4</sub>.

When expressed on a unit area basis, both confinement and mainly confinement systems had higher farm GHG emissions compared to unrestricted grazing systems and this was principally a function of differences in stocking rates. Total emissions (including SOC sequestration) were almost 50% lower for grazed systems compared to confinement systems, but there was a higher proportionate reduction in milk production. There was also a large impact of reduced input on a per unit area basis, with 38% lower GHG emissions per unit area (regardless of methodology) for the low input mainly confinement system compared to the high input confinement system. This was due to both lower stocking rates (lower CH<sub>4</sub> from enteric fermentation and manure management) and lower N<sub>2</sub>O emissions (no mineral fertiliser and lower organic N loading were applied on the low input system).

### **Impacts of production system on GHG emissions**

Confinement systems (total or partial) generally had a lower on-farm GHG emissions intensity due to higher milk production, and both lower CH<sub>4</sub> emissions and field N<sub>2</sub>O emissions per head. However, differences were reduced due to higher housing/storage emissions and a much reduced capacity for C sequestration. When emissions are expressed on absolute basis or per unit area basis, unrestricted grazed systems had much lower emissions (>40%) driven primarily by lower stocking rates. It was observed that when low input strategies were tested for both confinement (restricted grazing) and grazed systems that different results ensued. The low input, confinement system had much lower emissions per unit area, driven by reduced N input, lower stocking rates, higher feed intake and altered diet composition. However, this reduction in stocking rate resulted in a larger proportional decrease in milk production compared to N input reduction and resulted in a 7% increase in GHG emission per unit product compared to the high input confinement system. Furthermore, the low input unrestricted grazing system had 8%-10% lower emission compared to the high input unrestricted grazing system, regardless of the metric used. This was due to the fact that stocking rates and production capacity could be maintained in the low input system. These results demonstrate that tightening N surpluses in grazed systems can be achieved whilst maintaining production levels. As a result, differences were driven entirely by lower mineral fertiliser input. Such measures may be more difficult in high input confinement systems, where milk production is double that of grazed systems. In these cases, absolute emissions can be reduced, but a reduced N input may affect feed production and necessitate a reduction in stocking rate.

Next to N fertilisation, also lower stocking rate in itself may affect emissions in grazed systems. A lower enteric CH<sub>4</sub> kg<sup>-1</sup> milk was estimated with lower stocking density as a result

of a higher herbage allowance and intake by cows, resulting in a higher milk yield (simulation results for Irish unrestricted grazing systems; results not shown). Stocking densities were relatively high however, with 4.5 to 6.4 cow ha<sup>-1</sup>, and this result is probably not typical for lower levels of stocking density of 2.2 for the farm cases studied in the present study. The result is illustrative however for how level of intensity of farming may influence the effect of proposed mitigation measures, with cow performance becoming crucial.

The principal differences between the (partly) confinement and grazed systems were in terms of field emissions, which were almost 20% higher in unrestricted grazed systems, driven mainly by pasture paddock and range emissions. Similar large differences were observed for C sequestration rates. However, as these emissions are driven not only by N input but by abiotic factors such as soil climate characteristics as well, some of the largest discrepancies were observed between Tier 2 and Tier 3 for this cohort of emissions. It is clear from this study that assessment of soil emissions and soil carbon sequestration at the farm scale necessitates the generation of specific emissions factors based on local soil type, climate and land history. Also principle differences existed for enteric CH<sub>4</sub> emissions. The differences in CH<sub>4</sub> emissions among farm cases were smaller than those for N<sub>2</sub>O emissions due to the large contrast in levels of N fertilisation applied in these farm cases, but still of a similar order as the differences in C sequestration between confined and grazed systems. Differences in enteric CH<sub>4</sub> were related to the intensity of feeding and purchase of feed (concentrates and other non-forages) next to the change from grass silage to maize silage. The dietary effect of lower enteric CH<sub>4</sub> with the confinement systems existed for both the system with a high (most intensive) and low (most extensive) N fertilisation rate. From a nutritional point of view the dietary effects on enteric CH<sub>4</sub> are well controllable and reproducible, with the ruminant host regulating the rumen environment, probably far more reproducible and controllable than N<sub>2</sub>O emissions and perhaps C sequestration as well because of the profound impact and high variability of meteorological conditions on soil conditions and vegetation.

### **Predicted N<sub>2</sub>O emissions and C sequestration by process-oriented models**

The impacts of various mitigation strategies on soil N<sub>2</sub>O emissions and C sequestration were similar when modelled by either DNDC or PaSim. The impact of N fertilisation rate, for example, was similar both in terms of absolute emissions and the relationship between N application rate and N<sub>2</sub>O emissions. Both models indicated an exponential or linear response, dependent on soil type and/or climate. The DNDC model outputs suggest that the difference in response between sites was due to shifts in the N<sub>2</sub>/N<sub>2</sub>O ratio in response to N application rate. Also similar responses to percentage of legume in pasture were simulated, but these were observed to diverge from observed values due to the fact that neither model simulated the distribution of available N between the grass and legume. As PaSim has a more detailed plant physiology model, it is likely that a more accurate simulation of the impact of legumes may be possible whereas it is unlikely DNDC will adequately simulate these responses. The simulated outputs of the impact of stocking rates were similar for N<sub>2</sub>O emissions. However, DNDC predicted an increase in SOC with increased stocking rates up to 2.9 LSU ha<sup>-1</sup>, whilst there was no observed increase with PaSim. The differences may have been due to soil type differences in the farm cases simulated, but a more likely cause is the difference in methods to model C flows between labile and recalcitrant C pools and/or the proportion of C respired from each pool upon C input to the system. The C inputs from grazing led to a small but significant increase in recalcitrant pool carbon over time according to DNDC simulations. Differences may also be due to overestimation of grazed C return in DNDC as PaSim simulates grazed offtake of vegetation more accurately. Inhibitors were simulated in DNDC but not in PaSim, as DNDC has a highly detailed soil nitrification/denitrification sub-model and rates of microbial N nitrification can be altered.





In summary, both models had similar performance. The PaSim model may be more suitable for simulating mitigation strategies that primarily have an impact on vegetation whilst DNDC may be more suitable for simulating strategies that have a larger proportionate impact on soil processes.

### **Effect of adaptation on mitigation potential**

Modelling results obtained with PaSim for contrasting grassland systems, alternative management options and different agro-ecological zones in Europe indicate that decreasing the N fertilisation rate and stocking density of grazing animals can be valuable options to reduce emissions of both N<sub>2</sub>O and CH<sub>4</sub>. Reduction of N fertilisation can be accompanied by introduction or increase of the proportion of legumes in the grass/clover sward with an upper limit of about 20% to avoid the triggering of N<sub>2</sub>O emissions due to symbiotic N fixation that may occur at higher proportions. Legumes also contribute to N<sub>2</sub>O emissions in a number of ways, e.g. atmospheric N fixed by legumes can be nitrified and denitrified in the same way as fertiliser N, thus providing a source of N<sub>2</sub>O. Symbiotically living Rhizobia in root nodules are able to denitrify and produce N<sub>2</sub>O.

With variability in climatic conditions, the effect of a reduced N fertilisation on grass quality may be highly variable. Variation in grass quality (nutritive value as well as chemical composition) affects ruminant performance and enteric CH<sub>4</sub> emissions, which hence can be significant trade-offs that have to be accounted for when estimating mitigation potential. The emissions of GHG were further shown to be highly affected by the variability of climate conditions, indicating that grassland sites may become sources of C losses (or may reduce their sinking rates) in arid years, and this may occur regardless of the management type, which confirms the difficulty to sequester C in grassland soils by simply changing management practices.

Future climatic change will likely result in altered long-term climatic trends and also in both increased intra-annual climatic extremes and/or shifts in seasonality. An initial analysis of the impact of long-term weather trends and increased seasonal extremes in weather has been carried out. Initial impacts on net ecosystem CO<sub>2</sub> productivity (NEP) and soil organic carbon sequestration have been studied for temperate Atlantic regions. Long-term trends indicate 15% higher winter rainfall and 15% lower summer rainfall and a 1.5 – 2.0 degree Celsius increase in temperature, using outputs from the RCP 8.5 pathway (Moss *et al.*, 2008) and simulation of climate with a horizontal resolution of 0.035° (~4 km) using version 4.8 of the COSMO-CLM (Steppeller *et al.*, 2003). Results indicated a small decrease in NEP and SOC levels due to a proportional increased ecosystem respiration (primary productivity also increased). However, if weather volatility was introduced into the weather data, a maximum 25% reduction in NEP and 30% reduction in SOC sequestration were observed for grassland systems. Increased precipitation at higher latitudes may also increase N<sub>2</sub>O emissions. Mitigation strategies such as reduced stocking densities, grass mixtures and legume introduction may have a large impact into the future. Similarly, there may be reduced efficacy in Mediterranean regions, due to moisture deficits (but higher leaching if deluges follow drought). These impacts will be modelled further in the remaining months of the AnimalChange project. Previous studies on the 2003 European summer heat wave, have shown a 30% decrease in primary productivity and yields of both croplands and grasslands. As a result, grasslands were converted from a CO<sub>2</sub> sink to a source, releasing four years of C sequestration during this event (Ciais *et al.*, 2005). These outcomes emphasize the profound impact climate change may have on the GHG emission levels and the feasibility and efficacy of on-farm GHG mitigation measures

### **Effect of mitigation on adaptive capacity**

The implementation of mitigation management options should preferably not have negative impacts on the adaptive capacity of farming systems. For instance, reducing the stocking density of livestock may make a substantial difference to the CH<sub>4</sub> emissions when feed intake and feed quality are strongly affected, but the economic loss of the enterprise can also be high. In contrast, partial substitution of fertiliser N inputs with biologically-fixed legume N may reduce N<sub>2</sub>O emissions through reduced reliance on fertiliser N, which is an advantage of N limited systems with reduced costs of mineral N input. The presence of clover in grassland mixtures, while contributing considerably to the reduction of N fertiliser application needs, can help establishing a baseline in terms of adaptive capacity because it also improves the nutritional value of the forage (higher crude protein content than grasses) and thus animal production. Moreover, there is evidence that the presence of legumes in the sward tends to offset the effect of drought in grassland productivity in Europe (deliverable D4.1). Whether mitigating GHG emissions also serves the adaptive capacity of the farming system needs to be investigated, taking into account the consequences for intensity and profitability of farm production.

### **Integral assessment of emission factors by combining process-oriented models**

One of the important aspects of the present investigation was to identify how to pass variables from one process-oriented model to another. At the pasture-animal interface, the variables included for example crude protein. Pasture models are often constructed primarily as crop models, without regard to the subsequent use of the pasture produced for livestock feed. This emphasises the need for crop models that form the basis of ruminant diets (mainly, but not exclusively, grass and forage legumes) to include the simulation of these quality variables. In contrast, the output from detailed, complex digestion models may even contain more detailed data than actually needed for modelling C and N transformations in manure, necessitating an aggregation of data. On the other hand, complex digestive models do have the capacity to deliver important characteristics which determine N emission processes after excretion (e.g. urine volume and concentration of ammoniacal N) and during manure storage (manure volume, degradability of excreta components, acidity, ammonia emissions). Process-oriented models in principal represent the concentration-dependency of microbial processes, and hence it is of importance for any process-oriented model of ruminant digestion it also delivers information on the bulk of manure produced, the concentration of its constituents, and the characteristics of these constituents during manure storage and when applied to soil. This information was passed from the enteric fermentation model to the manure storage model and the soil emission model.

### **IPCC defaults versus emission factors derived by process-oriented modelling**

The differences in Tier 2 and modelled (Tier 3) N<sub>2</sub>O emission factors depended primarily on soil type and, to a lesser extent, climatic variation. Large underestimates in N<sub>2</sub>O emissions were observed when Tier 2 default emission factors were compared to modelled emissions from poorly-drained soils or soils with high clay content, such as gleysols. In contrast, Tier 2 and modelled emissions for loamy moderately drained soils were comparable, and overestimated (direct N<sub>2</sub>O emission) for free-drained sandy soils. Soil C sequestration for managed pastures was generally in good agreement between Tier 2 land-use factors and modelled outputs for temperate grasslands, although there was a small but consistent underestimate using Tier 2. However, there were large discrepancies on the impact of tillage on SOC loss with Tier 2 overestimating losses on sandy and loamy soils by 50-70%.

Also Tier 3 and Tier 2 estimates for enteric CH<sub>4</sub> differed substantially, and for various mitigation measures the Tier 3 simulates variation that in principal cannot be covered by a



the default used in the Tier 2. Examples of what the Tier 2 does not cover are the effect of N fertilisation on grass sward and grass quality, the exchange of grass versus maize, the consequence of nutrient limitation of milk production, and the supplementation of ruminant diets. A diversification of the Tier 2 approach is adopted in PaSim (based on grass digestibility), which probably does not reproduce the large variation encountered in confinement systems. However, as the use of PaSim is restricted to pasture-based extensive livestock systems, the approach taken for enteric CH<sub>4</sub> may suffice for the aim of PaSim to cover the consequences of variation in pasture quality.

The Tier 3 for enteric CH<sub>4</sub> is able to deliver details on excreta characteristics, enabling a diversification of the consequences of nutritional measures on enteric CH<sub>4</sub> on the one hand, and consequences on excreta components, their degradation characteristics and excreta volumes on the other hand. The latter serves as input for the process-oriented manure model and enables the evaluation of impact of nutritional measures and forage qualities on CH<sub>4</sub> emission from manure. The manure model estimated CH<sub>4</sub> emissions of similar size as the Tier 2 estimates, but also simulated substantial variation not covered by Tier 2 approach for enteric CH<sub>4</sub> and manure CH<sub>4</sub>. The impact of diet quality and feed intake level on the quantity of manure and the manure characteristics are not well covered by the Tier 2 approach.

### **Modelling results at animal/manure/field scale versus modelling results at farm scale**

There will be an interaction between all field-oriented mitigation measures and cattle-oriented measures that relate to the different mixtures of grass herbage, grass silage, maize silage, and supplemental such as with concentrates, by-products and other products. The field-oriented measures will all to a greater or lesser extent change the quantity and quality of feed available to the livestock, and will therefore affect intake, enteric CH<sub>4</sub> production, and emissions from the manure of CH<sub>4</sub> and N<sub>2</sub>O.

The current simulations consider the transformations of C and N in pasture land, the consequences of feeding forage to cattle on C and N transformations in the digestive system and the consequences of excretion on C and N transformations in stored manure. The emission of NH<sub>3</sub> in animal housing is an important process which affects indirect N<sub>2</sub>O emissions and amount of manure N applied to soil, but its contribution and variation remained underexposed in the present study. Furthermore, the chain of model simulations was terminated with the stored manure, so it does not include the consequences of applying the manure to pasture *i.e.* the cycle was not completed. To have completed the cycle would have required that the outputs from the manure model were adapted to conform with the input requirements of the pasture model; this would not have been insurmountable. However, to be internally consistent, it would also have been necessary to embed the three models within a framework capable of iterating through the models until the system reached equilibrium. This was not done in the present study, but will be addressed with the farm-oriented modelling efforts within AnimalChange.

### **Relevance of process-oriented modelling for estimation of emission factors**

Higher Tier approaches and the use of process-oriented modelling to generate emission factors is invaluable for investigating the impact of mitigation strategies on GHG emissions. The impact of strategies on underlying gross component fluxes can be assessed as well as the net effect on farm emissions. A more textured analysis is also achievable, for example, for example when analysing the impact of changes in feed quality and quantity on C input or C sequestration and on the size of labile or recalcitrant C pools in stored manure and soil. Higher Tier approaches also allow for differential soil and/or climatic impacts to be assessed. As such, these models can reflect emissions (and the impact of mitigation strategies) more accurately at a local basis (field, herd or individual farm). As a result, they are more useful than defaults to reflect the differences between farms in the same geographical region, or to

assess the impact of abiotic factors, such as soil, climate on farm emissions or to assess feeding aspects of farming in different regions. They are useful for comparing different farming systems, but only if enough activity data and other input data are available such as detailed information on feeds, housing, soils, manure and the local climate. As a result, Tier 2 emissions factors may be more appropriate and easy to use for comparison of different production systems across different regions as variation in soil type and climate is smoothed across these farm systems and only variation in input rates is assessed. However, the ultimate value of process-oriented models may be estimate and diversify the lower Tier emission factors. These models may be used to assess the proportionate impact of type and management of soils, crops, diets and manure on C and N emission factors and, in turn, generate a matrix of emission factors based on, for example, N input rate, N type and soil type.

The profound differences between emission factors according to IPCC Tier 2 methodology and predicted by process-oriented models indicates that Tier 2 methodology should be abandoned when investigating consequences of variation in farm management and farming conditions on GHG emissions. Process-oriented models represent the underlying mechanisms of processes in e.g. the gastrointestinal tract of an animal, in soils, and in manure storage. It is hence the mechanistic nature of the models which distinguishes them from more empirical approaches, and which enables them to introduce a logic to studies that identify changes in GHG emission under various management and environmental conditions. This logic is lacking when using predictive equations and default emission factors which have a purely empirical basis. Default emissions factors must be seen as representation of empirical data and their applicability is hence restricted to the conditions that hold for the data sets that were used to derive them. This means that more extreme conditions are likely not covered by these empirical approaches if those extreme conditions were not part of the data set. Also, empirical approaches may explain observed variation less well, as for example demonstrated by Ellis *et al.* (2010) for enteric CH<sub>4</sub>. Therefore, even though process-oriented models normally require more data inputs which actually may contribute to uncertainty in model outcome (Kros *et al.*, 2012), they do have the potential of higher precision under a wide range of conditions, and to deliver logic and coherence in the large set of assumptions made with inventories.

## 8. Concluding remarks

This report provides estimates by process-oriented models of mitigation and adaptation options calculated with PaSim, DNDC, the Dutch Tier 3 for enteric CH<sub>4</sub> in dairy cows, and a newly developed manure storage model. In evaluating effects on GHG emissions from farms, sufficient detail and accuracy appears necessary for accuracy of emission factors for individual sources and sinks. Sources and sinks of GHG have to be quantified in an integrated manner, meaning that they cannot be treated to be independent from each other for reasons of convenience, but that they have to be estimated taking the underlying processes and mechanisms into account. Interactions occur between various aspects of farm management as well as climatic factors, and between various emission sources (mainly following non-linear relationships). Gathering information on a more theoretical basis is essential to provide an understanding of the mechanisms involved and to introduce logic in how various emission sources can be related to some key farm factors (e.g. level of N fertilisation, crop yield, and animal feed intake and productivity). It can be studied how the changes in emission sources are interrelated, and how mitigation options are affected by climate scenarios that would require adaptation of farm management.

There is no major limitation for a combined use of the models as inputs and outputs could be made compatible and exchangeable among the models, and in this respect models could communicate. A combined modelling effort was undertaken for a selection of dairy farms that contrasted in production intensity (milk production per ha and stocking density), intensity of feeding and intensity of N fertilisation. This selection of contrasting farm types delivered insight in how trade-offs or synergies of distinct on-farm sources relate to dairy farm characteristics and local conditions.

The options described in this report are options at the field and the animal scale. For a summary of the detailed effects on emission factors the reader is referred to Tables 3.1 for the field scale and Table 4.1 for the animal scale. It is important to have a clear understanding of the effects of possible options at these scales, since it is the interface between biophysical processes and human intervention, and the scale where farmers make their day-to-day decisions. However, it is also important not to forget the regional and global effects, since decisions at the scale of field and animal may affect the regional and global scale, and vice versa. For this reason, it is important that the present simulation results become upgraded to a higher level. This should be done by taking into account the precise background and causes of the variation in emission factors as a replacement for the use of the rather generic IPCC values which have, in principle, not been generated for case-specific use.



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