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Modelling farm and field emissions in LCA of farming systems: the case of dairy farming

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ABSTRACT

The generic nature of LCA with a wide coverage of environmental processes and impacts, calls for a simplified environmental modelling. This is notably the case for agricultural and food systems, where often generic emission factors (EFs) are used such as those derived from the IPCC guidelines. These EFs reflect country averages or a global situation, but are not suited for comparing different farming systems, since they do not take into account specific management, climate, and soil parameters and therefore do not distinguish between different farming systems. Moreover, emission models in LCA tend to have varying degrees of detail, which means that climate, soil and farming practices are reflected for some emissions and processes, while ignored for others. In pasture-based livestock systems, direct emissions from animals and pastures tend to dominate many of the environmental impacts such as climate change, eutrophication or acidification. The objective of this study is to improve emission modelling for nitrogen (N) of pasture-based dairy systems in comparison to intensive dairy production using high amounts of concentrate feed.

The Swiss Agricultural Life Cycle Assessment (SALCA) method uses a nutrient balance model of a herd to calculate emissions from animal husbandry and nutrient excretion. The effects of feed intake, feed quality, and different levels of productivity on emissions and environmental impacts can thus be represented. To better consider the specific aspects of pasture-based dairy systems, the modelling of N excretion in urine and dung and subsequent N emissions is revised. The excretion of N in urine and dung is modelled as a function of N concentration in the diet. Subsequently different emission factors for N_2O, NH_3 and NO_3 are applied from urine and dung and for N_2O emissions from organic fertilisers. Furthermore, a correction factor for N_2O emissions after the application of N fertilisers is calculated as a quadratic function of the N rate. The revised emission models can result in changes of over 50% compared to use of generic EFs and should better reflect the N emissions from pasture-based dairy systems.

Keywords: dairy production, nitrogen emissions, nitrogen excretion, pasture

1. Introduction

The nitrogen and the carbon cycles are key drivers for the environmental impacts of agricultural systems (Williams *et al.*, 2010), since they are major determinants of several impact categories like e.g. the global warming, the eutrophication and acidification.

Grassland-based dairy systems are important examples of agricultural systems, where the nitrogen and carbon cycles play a key role (Ledgard, 2001). Methane from enteric fermentation and manure management, nitrous oxide from manure management, and deposition to soils, ammonia emissions related to animal husbandry and manure management as well as nitrate emissions from grazing and spreading manure are key drivers for the environmental impacts of dairy farming (Guerci *et al.,* 2013). The emissions of the greenhouse gases (GHG) methane and nitrous oxide are typically determined by simple EFs in LCA. EFs were developed for the use in national GHG inventories (IPCC, 2006), designed for the accounting at the national scale. Using the same EFs for specific farming systems might be inappropriate, since the specific conditions of the analysed systems are not taken into account

(Peter *et al.*, 2016). Therefore, the EFs and models often used in LCA might not adequately represent the emissions in real systems, since they do not reflect specific aspects of the investigated systems. Improving nitrogen emission models was identified as a top priority for the improvement of LCA of livestock systems (Cederberg *et al.*, 2013). The default EFs for nitrous oxide (N_2O) from agricultural soils from IPCC (2006) are unspecific and for example do not distinguish between different types of fertilisers or between excretion of urine and dung from grazing animals.

A further challenge is the varying level of detail for several emission pathways. This can be clearly seen for the different nitrogen losses in the form of ammonia, nitrous oxide and nitrate. While for ammonia (EEA, 2013; HAFL, 2013) and nitrate (Richner *et al.*, 2014) relatively detailed EFs and models are available taking into account the type of fertiliser, the timing of application, the soil and climate factors, nitrous oxide emissions from fertilisation are mainly calculated from the amount of N applied (IPCC, 2006). Therefore, variations in climate and soil conditions as well as mitigation measures cannot be evaluated properly for all emissions.

Pasture-based dairy systems differ from systems with high use of concentrate feed in many respects: animal husbandry (barn feeding vs. grazing), milk yield per cow, type of feed, manure management (manure spreading vs. excretion on pasture), and type of land use (grassland vs. arable land). Ideally, the EFs and models should reflect these aspects, in order to achieve a differentiated analysis. This paper presents the approach followed in the Swiss Agricultural Life Cycle Assessment (SALCA) method to model emissions from animal husbandry (section 2.1) and the recent improvements made in order to achieve a more adequate analysis of pasture-based dairy systems, with focus on the N modelling and emissions (sections 2.2 to 3.5).

2. Modelling emissions from animal husbandry

2.1 Overview of the concept

Calculations for emissions from animal husbandry and nutrients (N, P, and K) excreted by the animals are performed by a nutrient balance model of a herd in the SALCA method (Fig. 1). It takes into account the specific feed intake and quality, the export of animal products, changes in live weight, and emissions. The effects of feed intake, feed quality, and different levels of production on emissions and environmental impacts can thus be represented.

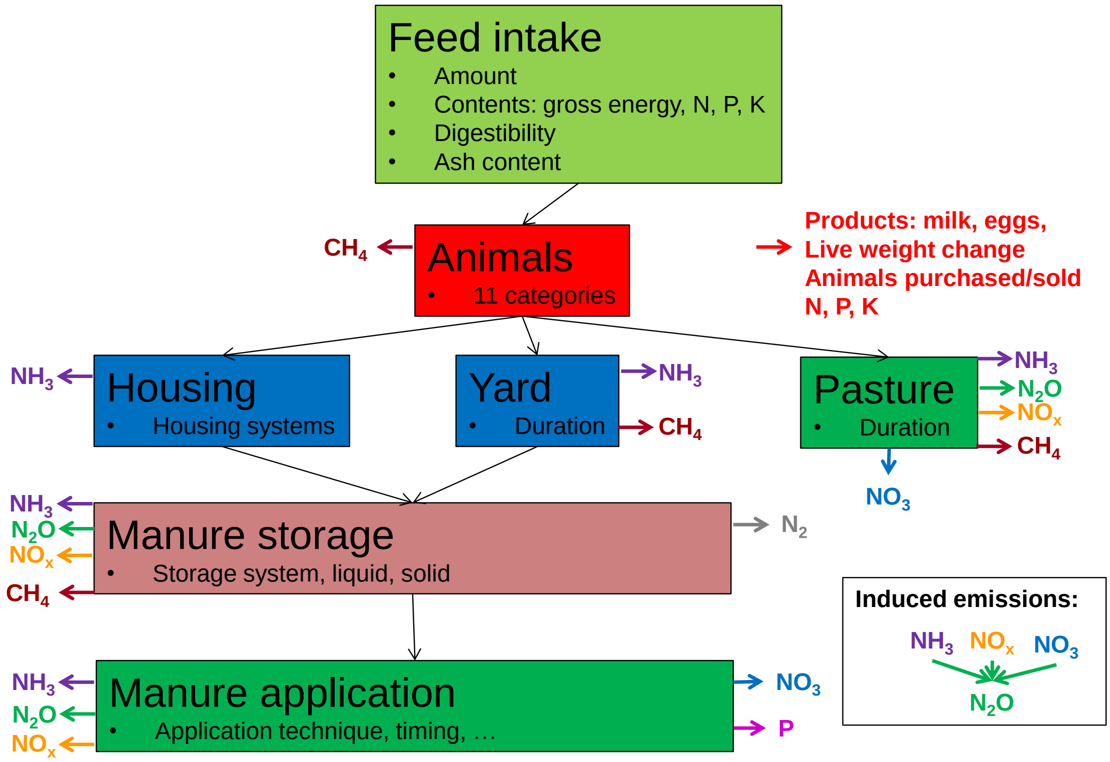

Figure 1: Flow diagram to model nutrient balances and emissions from animal husbandry (SALCA-Animal). Nutrient flows and emissions related to application of other fertilisers than animal manure are calculated separately and not shown in the figure.

The following EFs and models are used in SALCA to model emissions from animal husbandry: Agrammon (HAFL, 2013) or EEA (2013) Tier 2 for ammonia depending on the context of the study, IPCC (2006) Tier 2 factors for methane complemented by Kirchgessner *et al.* (1995) for dairy cows, IPCC (2006) Tier 2 factors for N_2O from manure management, EEA (2013) for NO_x and N_2 emissions, and Richner *et al.* (2014) for NO_3 leaching.

Details are given in Bystricky *et al.* (2014) and Nemecek *et al.* (2015). In the following sections the adaptations of the models are discussed to better account for the specific aspects on pasture-based systems. The focus in this paper is on the refinement of N emission modelling from grazing cow systems.

2.2 Nitrogen partitioning between urine and dung

Most emission models used in LCA do not take into account the partitioning of excreted N between urine and dung, e.g. EEA (2013) calculate N excretion in a balance model and assume a fixed proportion of TAN in total N of 60%. However, the proportions of N excreted in urine and dung vary considerably (Bracher and Menzi, 2015; Luo and Kelliher, 2014) and are influenced by many factors. Digestible N enters the cow's metabolism and is either used to produce milk or to form body tissue; the remaining N is excreted in urine. The non-digestible fraction of N is excreted in dung. The proportion of N excreted in the urine increases strongly with increasing N intake and increasing N concentration in the diet, while the proportion of N excreted in the dung changes only little in function of the above-mentioned parameters (Luo and Kelliher, 2014). Taking these relationships into account is crucial for emission calculations. N in urine is mostly composed of urea (73% according to Selbie *et al.* (2015)), the rest consists of various soluble N compounds. In fresh cattle dung, 99% of the N is organically bound, which is partly water soluble (fresh cattle dung contains 21.5% soluble N compounds (Kirchmann and Witter, 1992)). Due to these different compositions of urine and dung, the emissions of N_2O, NH_3 and NO_3 are much higher from urine than from dung.

Luo and Kelliher (2014) proposed relationships between N concentration in the diet and the N excretion in urine. A comparison of their numerical relationship to several experiments carried out in Switzerland however showed relatively large deviations particularly for low protein diets. Therefore the relationship was re-estimated for use in Swiss dairy systems using data from 14 studies published between 1993 and 2014 (total of 420 data points, personal communication of A. Bracher, Agroscope, May 2016 and Bracher and Menzi (2015), see Fig. 2).

A linear regression was calculated for lactating cows as

%Nurine = 4.7+20.7*%Ndiet (n=389, r^2=0.65) (eq. 1)

where

%Nurine = N excreted in urine [g N/day]/total N excreted in urine and dung [g N/day]*100

%Ndiet = N concentration in the diet [g N/kg dry matter]*100.

A linear relationship is plausible only in a certain range. From the analysis of moving averages it was concluded that average values below 30% and above 70% are unlikely and the linear function was therefore truncated at these limits (Fig. 2). It should be noted that for individual cows and in specific circumstances higher and lower values can occur. Dry cows were not included in the regression. However, dry cows have generally low %Ndiet and the corresponding values were around 30%; therefore the truncated function can also be applied to dry cows.

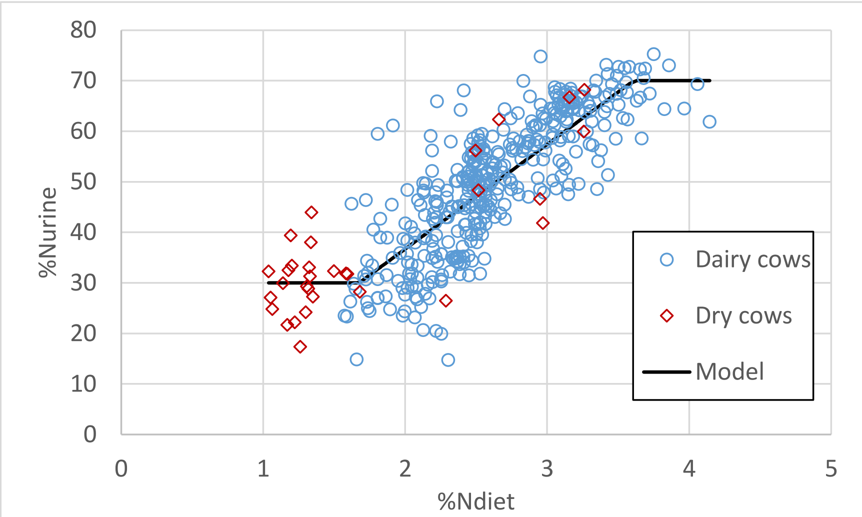

Figure 2: Proportion of N excreted in urine (%Nurine) as a function of the N concentration in the diet (%Ndiet) on a dry matter basis.

This relationship is used for the subsequent modelling of N_2O (sections 3.1 and 3.3), NH_3 (section 3.4) and NO_3 (section 3.5) emissions. It is also used to estimate the proportion of TAN in the total N in the excrements as a starting point for the modelling of N dynamics (mineralisation, immobilisation) and N emissions during manure storage and subsequent manure spreading.

3. Modelling N emissions from manure management and fertiliser application

3.1 Direct nitrous oxide emissions from grazing cattle

N_2O emission rates from cattle excrements deposited on pasture are generally higher than N_2O emissions from applied fertilisers (Kelliher *et al.*, 2014). IPCC (2006) therefore proposed a default EF of 2.0% for grazing cattle excreta N compared to 1.0% for N fertilisers. The reason is that cattle excreta have a high N concentration, which corresponds on average to a local application rate of 613 kg N/ha within urine patches (Selbie *et al.*, 2015). The local N concentration is too high, so that N cannot be taken up by the vegetation quickly enough, which results in higher losses compared to the low N rate from an even distribution of N fertilisers.

Numerous experiments have shown that N_2O emissions from urine are considerably higher than from dung. N is organically bound in dung and needs to be mineralized first, before denitrification and nitrification can start, and N_2O can be formed. This leaves more time to the vegetation to use the N, resulting in lower emission rates. Kelliher *et al.* (2014) analysed data from 185 experiments carried out in New Zealand between 2000 and 2013. In the lowland soils the mean EFs of 1.16% were found for cattle urine and only 0.23% for cattle dung. For the hill areas (with slopes >12%), significantly lower EFs were found (Luo *et al.*, 2013). Higher EFs for urine N as compared to dung N were also found by Bell *et al.* (2015) for Scotland, Mori and Hojito (2015) in Japan, Sordi *et al.* (2014) in Brazil, and Rochette *et al.* (2014) in Canada.

In the New Zealand's GHG inventory, national Tier 3 EFs of 1.0% and 0.25% are used for cattle urine and cattle dung excreted on pasture, respectively (MfE, 2015). These values are considerably lower than the IPCC default value for grazing cattle, which is 2.0% for all excreted N. Lower EFs can be observed for urea, where the IPCC default EF is 1.0%, while the mean EF determined in New Zealand was only 0.48% (Kelliher *et al.*, 2014). The lower N_2O emissions in New Zealand are mainly explained by the different soil types, which are often of volcanic origin and tend to be coarse-textured and well-drained, where optimal conditions for the production of N_2O are less frequent. De Klein *et al.* (2014) found lower EFs on free draining soils as compared to poorly drained soils.

Therefore the average value of 2.0% for emissions from urine is used in SALCA from a meta-analysis (Selbie *et al.*, 2015), while for dung a value of 0.5%, i.e. 4x lower is applied, thus keeping the same relationship as in New Zealand's national GHG inventories. If these EFs are combined with the proportions excreted in urine and dung from eq. 1, depending on the diet the resulting EF for N_2O is 23-53% lower than the IPCC default, previously used in SALCA (Fig. 3).

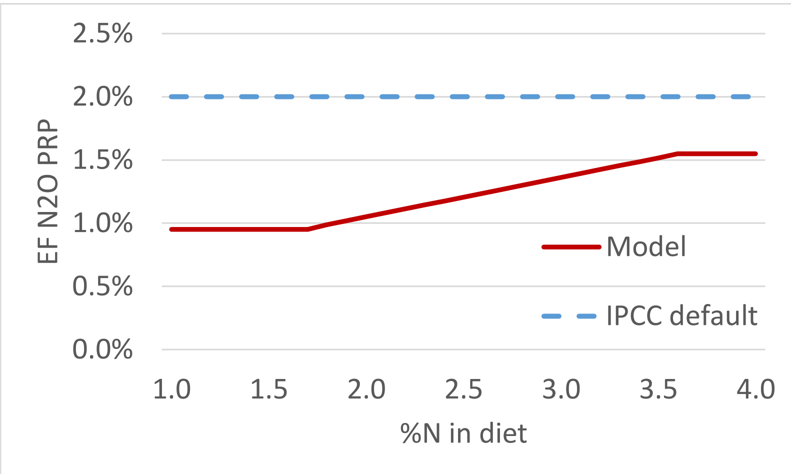

Figure 3: EF for N_2O on pasture (EF N2O PRP) compared to the IPCC default (Tier 2).

3.2 Direct nitrous oxide emissions after application of N fertilisers at different rates

Higher rates of N application tend to result in higher emissions of N_2O (Bouwman *et al.*, 2002). At higher N rates, the vegetation might not be able to take up the N quickly enough. Shcherbak *et al.* (2014) analysed 78 studies with several levels on N application and found a non-linear response of N_2O emissions. Using the relationship derived for all experiments except four outliers, a correction factor ($CF_{N2Orate}$) can be derived as a quadratic function of the N fertiliser rate:

$$CF_{N2Orate} = 0.1*(1.036/N_{fert}+6.42+0.0244*N_{fert}) \qquad \text{(eq. 2)}$$

where

N_{fert} = N fertiliser applied [kg N/ha/year]

The correction factor equals 1 at a rate of 146 kg N/ha/year, which corresponds to typical N doses. Lower N rates lead to reduced N_2O emissions as compared to the IPCC default value, while higher N rates give greater emissions (Fig. 4).

This correction factor is multiplied with the EFs for N_2O for mineral and organic fertilisers except for excreta from pasture (EF N2O PRP, see section 3.1). The reason is that in the latter the N application rate is already taken into account in the higher EF for N in urine as compared to N fertilisers. Interestingly, the $CF_{N2Orate}$ for 613 kg N/ha/year (the average urine N rate, see section 3.1) yields a factor of 2.1, a value which is very close to the EF_{N2O} for cattle urine of 2.0%.

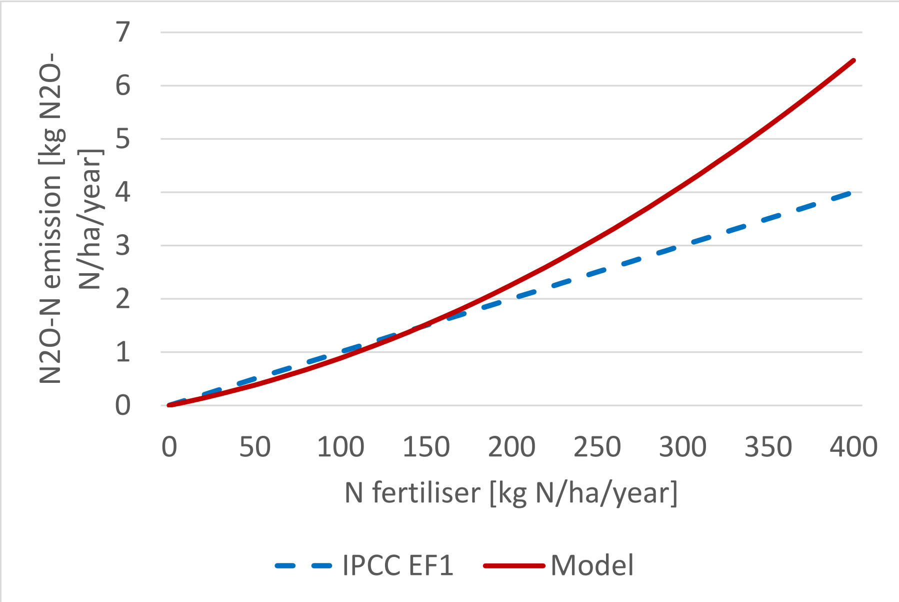

Figure 4: Emissions of N_2O from fertiliser applications with correction for the N application rate compared to the IPCC default (Tier 1).

3.3 Direct nitrous oxide emissions after application of organic fertilisers and manure

Since only a part of the N contained in organic fertilisers and manure is readily available, the emission rates can be expected to be lower than from mineral fertilisers (Bouwman *et al.*, 2002; Stehfest and Bouwman, 2006). However, the emissions depend on many factors such as type of organic fertiliser, soil, climate, conditions of application, etc. (Lesschen *et al.*, 2011) so that generic conclusions are difficult to draw (Thangarajan *et al.*, 2013). We opted therefore for the application of a simple robust model.

The TAN of organic fertilisers is known either from the N balance model (Fig. 1) in the case of farmyard manure, from standard tables or specific analyses. The default EF1 for N fertilisers (1.0%) will be used for the mineral part of N (TAN) as for the mineral fertilisers. For the organically bound fraction (total N - TAN), a factor of 0.25% will be used, i.e. a value 4x lower, thus keeping the same relationship as between urine and dung excreted on pasture. Note that the SALCA-animal model presented in Fig. 1 includes also mineralisation of N in liquid manure and immobilisation of N in solid manure.

3.4 Ammonia emissions from grazing cattle

Ammonia emissions from dung excreted on pasture can be considered as negligible, since almost all N is organically bound (Kirchmann and Witter, 1992). For urinary N, Selbie *et al.* (2015) give an average emission rate of 13% (median 12%; kg NH_3-N/kg N in urine) from a meta-analysis. Most of the measurements were done in summer, when ammonia volatilisation is highest. A seasonal variation can be derived as 8% in spring, 15% in summer, 9% in autumn and 7% in winter (Selbie *et al.*, 2015).

3.5 Nitrogen leaching from grazing cattle

Nitrate leaching shows a seasonal pattern, with highest leaching rates in autumn and winter and lowest in spring (Selbie *et al.*, 2015). From a meta-analysis the average leaching rates from urine patches were estimated as 17% in spring, 16% in summer, 24% in autumn and 20% in winter (Selbie *et al.*, 2015). N leaching from dung was measured only in a few studies and the results indicate much lower leaching rates than for urine, but do not allow a robust estimate of the leaching rate. As only 21% of the N is water soluble (Kirchmann and Witter, 1992), much lower leaching rates can be assumed. Therefore the above leaching rates for urine patches were extrapolated by multiplying by 0.2 in the case of N in dung. Most data on N leaching stem from lysimeter studies. In most of these experiments, the whole area of the lysimeter was wetted with urine. Buckthought *et al.* (2016) has shown that not only the plants growing in the wetted area use the N in the urine patch, but also the surrounding vegetation. 20% of the urinary N applied in spring was taken up by the surrounding vegetation. The authors conclude that many lysimeter studies might overestimate leaching under real conditions. Therefore, a correction factor of 0.8 will be multiplied by the calculated leaching rates for grazing in spring and 0.9 for grazing in summer to determine the effective N leaching, summarised in Table 1. No correction will be applied to autumn and winter application, since the recovery of N by the plants is low in these seasons. Higher stocking rates lead to more frequent overlaps between excreta, which increases the risk of N leaching. E.g. Ledgard *et al.* (2015) has shown increasing leaching rates with increasing N excretion from grazing animals. Such relationships could be taken into account in future model improvements.

Table 1: N leaching factors as a function of the season.

	Spring	Summer	Autumn	Winter
N leaching from urine [kg N leached/kg N excreted]	13.6%	14.4%	24.0%	20.0%
N leaching from dung [kg N leached/kg N excreted]	2.7%	2.9%	4.8%	4.0%

4. Discussion, conclusions and outlook

The proposed changes in the N emission modelling better take into account specific aspects of pasture-based dairy systems. In particular the effect of different diets on the emissions of N_2O, NH_3 and NO_3 is accounted for. Grazing dairy cows generally have diets with high protein contents, which leads to a higher proportion of N excreted in urine and subsequently higher N emissions. Furthermore, the N concentration of the diet depends on the composition of the feedstuffs and is related to the milk yield. In this context it is important to distinguish between diets during the grazing season and winter diets instead of using average annual diets. However, cows with low N concentrations could also have low milk yields, so that the emissions and impacts per kg milk have to be evaluated through the LCA of the whole dairy system in the next steps.

The adapted models will be tested in the project "Optimisation of grassland-based dairy systems with grass cutting and barn feeding" comparing three production strategies for dairy cows in

Switzerland: 1) full grazing, 2) grass cutting and barn feeding low amounts of concentrates and 3) barn feeding using moderately high amounts of concentrates.

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