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Greenhouse gas emissions from dairy production – assessment and effects of important drivers

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List of abbreviations

APRF(s)	(Regionally) alternative produced protein-rich feedstuff(s)
APRM(s)	Mixture(s) of APRFs
C	Carbon
CaO	Limestone
CF	Carbon footprint (Introduction; publication 3, section 2.3; Overall discussion and Conclusions)
CF	Dietary crude fibre (publication 1, section 2.1)
CH_4	Methane
CO_2	Carbon dioxide
CO_2-eq	Carbon dioxide equivalents
C_{org}	Soil organic carbon, humus
CP	Dietary crude protein
DDGS	Distillers' dried grains with solubles
DM	Dry matter
EE	Ether extracts
FB	Faba beans
Gg	Gigagramme
GHG(s)	Greenhouse gas(es)
GHGE(s)	Greenhouse gas emission(s)
GMO	Genetically modified organism
Gt	Gigatonne
GWP	Global warming potential
ha	Hectare
iLUC	Indirect Land Use Change
IPCC	International Panel on Climate Change
K_2O	Potassium oxide (for content of potassium in fertilisers)
LC	Lucerne cobs
LCA	Life Cycle Assessment
LU	Land-use
LUC	Land use change
LULUC	Land-use and Land use change
Mg	Megagramme (tonne)

MJ	Megajoule
N	Nitrogen
N_2O	Dinitrous oxide
NEL / NE_L	Net energy lactation
NfE	Easily soluble carbohydrates (N-free extracts)
P_2O_5	(Di)phosphorpentoxid (for content of phosphorus in fertilisers)
ppb	Parts per billion
ppm	Parts per million
PS	Production system(s)
RSC	Rapeseed cake
RSME	Solvent-extracted rapeseed meal
SBME	Solvent-extracted soybean meal
SSC	Sunflower seed cake
SSME	Solvent-extracted sunflower seed meal
uCP	Crude protein utilizable in the duodenum
UDP	Rumen-undegradable protein

Abstract

This thesis presents new findings on greenhouse gas emissions (GHGE) of dairy production, on mitigation effects of a substitution of specific feedstuffs and on GHGE from land use and land use change (LULUC).

An extension of system boundaries for calculation of GHGE, especially including emissions related to LULUC, resulted in advantages for dairy production systems (PS), which hardly utilise the LULUC-burdened soybean meal. Within different production methods, PS with a higher milk output generally showed better results for GHGE per kg of milk produced as compared to PS with a lower milk output. Nevertheless, the latter showed clearly better results for GHGE per ha of land used. The regional location of the farm with its impact on forage quality or milk yields was identified as another very important driver for emission loads per kg of raw milk. These results emphasize the importance of a complete life-cycle assessment in the evaluation of impacts that dairy PS have on the climate.

Furthermore, the consequences were estimated of a substitution of protein feedstuffs which are specifically loaded with high GHGE from land use change (LUC). Highest GHGE were found for extracted soybean meal, mainly due to LUC-related emissions. Medium GHGE were found for distillers' dried grains with solubles, for cake and extracted meal from rapeseed and for lucerne cobs. Cake and extracted meal from sunflower seed as well as faba beans were connected to the lowest GHGE. Substituting soybean meal by nutritionally equivalent mixtures of alternative protein feedstuffs, resulted in an average reduction of GHGE of 42 %. Balanced mixtures of alternative protein feedstuffs may offer specific benefits, as they allow for a combination of desirable nutritional value and reduced GHGE.

With a focus on emissions from LULUC, this work suggests that GHGE which are included in assessments should be restricted to physically occurring fluxes of greenhouse gases and should exclude hypothetical sources. The results show that almost a quarter (23 %) of the increase in CO_2 concentration which occurred during the last 250 years originates from LULUC with a rather drastic impact of LUC occurring during the last few decades. Thus, CO_2 emitted from both soil and vegetation due to LULUC shall be considered within accounting periods of ten or

20 years. The length of these accounting periods is consistent with the timescale of soil carbon losses from isolated areas which remain in the atmosphere.

Zusammenfassung

Die vorliegende Dissertation hat Treibhausgasemissionen (THGE) der Milcherzeugung, Minderungspotenziale der Substitution von bestimmten Futtermitteln und THGE von Landnutzung und Landnutzungsänderungen (Land Use and Land Use Change, LULUC) zum Gegenstand.

Eine Erweiterung der Systemgrenzen in der Bewertung von THGE, im Besonderen um Emissionen aus LULUC, zeigte Vorteile für Produktionssysteme (PS) der Milcherzeugung mit geringer oder fehlender Verwendung von importiertem Sojaextraktionsschrot. Innerhalb vergleichbarer Produktionsmethoden weisen PS mit höheren Milchleistungen generell Vorteile bei produktbezogener Emissionsbewertung (pro kg Milch) gegenüber PS mit geringeren Leistungen auf. Umgekehrt zeigen letztere deutlich bessere Ergebnisse bezüglich der THGE pro Einheit beanspruchter Nutzfläche. Der Standort eines Betriebs wurde mit seinen Einflüssen auf Grundfutterqualität oder Milchleistungshöhe neben der Produktionsmethode als weiterer, sehr wichtiger Faktor für die Höhe von THGE pro kg Milch identifiziert. Die Ergebnisse zeigen für Lebenszyklusanalysen (Life Cycle Assessment, LCA) am Beispiel der THGE von PS der Milcherzeugung die hohe Relevanz einer breiten Systemgrenzensetzung und einer möglichst vollständigen Analyse aller Prozesselemente innerhalb des Bewertungsrahmens.

Des Weiteren wurden in der vorliegenden Arbeit die Minderungspotenziale einer Substitution von bestimmten Proteinfuttermitteln untersucht, die besonders hohe THGE von LULUC aufweisen. Höchste THGE zeigte dabei importierter Sojaextraktionsschrot, mittlere Emissionsbelastungen Trockenschlempe, Kuchen und Extraktionsschrot von Raps sowie Luzernecobs. Kuchen und Extraktionsschrot von Sonnenblume, sowie besonders Ackerbohnen wiesen geringste Emissionen auf. Eine Substitution von Sojaextraktionsschrot mit gleichwertigen Mischungen alternativer Proteinfuttermittel reduzierte die THGE durchschnittlich um 42 %. Ausgewogene Mischungen der alternativen Proteinfuttermittel haben besondere Vorteile aufgrund der Kombination einer günstigen Nährstoffzusammensetzung mit geringerer Emissionsbelastung.

Im dritten Teil zeigt die vorliegende Arbeit, dass die Bewertung von THGE auf physische, real existierende Stoffflüsse beschränkt und nicht auf hypothetische

Emissionsquellen erweitert werden soll. Knapp ein Viertel (23 %) des Anstiegs der atmosphärischen CO_2-Konzentration der letzten 250 Jahre ist auf LULUC-Emissionen zurückzuführen; dabei lässt sich ein besonderer Einfluss von Landnutzungsänderungen der vergangenen Jahrzehnte feststellen. Daraus wird gefolgert, dass LULUC-bürtige Kohlenstoff-Emissionen für eine Zeitdauer von zehn bis 20 Jahren nach einer Nutzungsänderung in THGE-Bewertungen aufgenommen werden sollen. Die abgeleiteten Empfehlungen für anzuwendende Anrechnungsperioden sind mit jenen für den atmosphärischen Verbleib eines Großteils des emittierten Kohlenstoffs bei Landnutzungsänderung von isolierten Flächen konsistent.

1 Introduction

Although relevant greenhouse gases (GHGs) from agricultural processes (carbon dioxide, methane, dinitrous oxide) account for a small proportion of the atmosphere's composition only, they show an important influence on climate. By reflecting the radiation emitted from the earth surface instead of releasing it to the outer space, the natural greenhouse effect makes life on earth possible at all (IPCC, 2001). In addition to natural greenhouse gas emissions (GHGEs), anthropogenic gases from burning and other use of fossil resources and renewable raw materials emit carbon dioxide (CO_2), methane (CH_4) and other (trace) gases relevant to climate and to ecosystems. CH_4 is furthermore emitted as a consequence of anaerobe decay processes from soils, especially in connection with the cultivation of rice (floated fields, deepwater cultivation), and digestion of feedstuffs, mainly roughages. An additional important source of anthropogenic GHGEs is land use change, contributing mainly to CO_2 from cleared or burnt biomass and a loss of soil organic matter during the subsequent agricultural use. Last but not least, dinitrous oxide (N_2O) which is assumed to represent the most important GHG from overall agriculture is produced in soils, especially after fertilising and in soils which are compacted as a consequence of the use of heavy machinery during cultivation and harvest (IPCC, 1997, 2006).

Effects of high anthropogenic GHGEs cannot be predicted easily, but are assumed to lead to abrupt changes of earth's climate within decades and centuries. Although a high variance exists for the estimated likelihood for certain occurrences, warming of the climate system is evident from observations such as the increase in global average air and ocean temperatures: eleven out of the twelve warmest years in the instrumental record of global surface temperature (i.e. since 1850; IPCC, 2007) occurred between 1995 and 2006. Furthermore, melting of snow and ice and rising global sea level also result from global warming.

These climate changes are assumed to be clearly linked to GHGEs as changes in atmospheric concentrations of GHGs – besides similar effects of aerosols, land cover and solar radiation – alter the energy balance of the climate system. The IPCC report deduces a high proportion of the observed increase in global average temperatures since the mid-20th century being related to the observed increase in GHGE concentrations (IPCC, 2007). These global GHGEs have remarkably grown since pre-industrial times due to human activities and have led to atmospheric concentrations of CO_2 (379 ppm) and CH_4 (1774 ppb) in 2005, which by far exceeded the natural variation within the last 650,000 years (IPCC, 2007). Hence, it is concluded that human activities since 1750 have resulted in warming.

Climate change is not only relevant to environmental systems, but also to socio-economical systems. Expected effects mainly include: a) Direct influence on human, animal and plant health, mainly by heat stress but also from flooding and severe storms. b) Indirect influence on health due to a change of water-, air- and food-quality and higher incidence of disease carriers, such as certain insects, and of water-borne pathogens. c) Change of ecosystems, including high rates of extinction of species connected with a multitude of effects on socio-economical systems. d) Declining yields of harvest in many – mainly subtropical and tropical – regions. e) Water shortage becoming worse in many regions, especially where water scarcity is already a problem. In particular, agricultural food production is affected by climate change, mainly due to higher maximum temperatures, a greater number of hot days and heat waves, imposing heat stress on plants and livestock. In many regions the decreasing frequency of cold days with frost will be advantageous for a number of plants, but will also result in a higher incidence of disease carriers and of plant pathogens. Intensive rainfalls which are assumed to come along with the climate change lead to soil erosion. For many regions of the world decreasing yields are assumed to be likely because of dry summers with draughts, water scarcity and increased frequency and extent of forest fires. Climate change is likely to have the greatest impacts and negative socio-economical consequences in developing countries. Hence, already existing problems of social inequity concerning access to food, fresh water and other resources will deepen (IPCC, 1996, 2001).

Agriculture is not only affected by climate change, but is also a source for anthropogenic GHGEs. Consequently, besides short-term partial adaption of agricultural management practices to climate change, agriculture needs to react on unfavourable future production conditions and strategically reduce its GHGEs.
Since the issue of GHGEs and climate change are intensely covered in the media, cattle and other ruminants are presented as main sources for GHGEs, but are

much less valued for their ability to convert roughage into milk and meat for human nutrition (Gill et al., 2011). Globally, agricultural livestock directly emits 9 % of total GHG (IPCC, 2007). Following the sectoral allocation of GHGE according to IPCC guidelines (e.g. IPCC, 2006), this 9 % include emissions from enteric fermentation, soils and manure management systems. Dairy cows and other cattle account for a high proportion of these GHGE. This applies to the global scale, but also to European alpine regions, including Austria. If emissions from other sectors concerning the whole supply chain of livestock products are incorporated, livestock's contribution to global emissions may be as high as 18 % (Steinfeld et al., 2006).

As opposed to the sectoral allocation of GHGE according to IPCC, the product's *Life Cycle Assessment* (LCA) covers the effects which are imposed on the environment through the supply of a product during its whole life cycle (UNEP, 2009). Accordingly, for livestock production steps such as use of (fossil) fuels and energy, transports, fertilizer production or deforestation and land use change before cultivation of feedstuffs are included into estimation of impacts on environmental systems in LCAs. This method of LCA was introduced in the late 1960ies to analyse the environmental impacts of alternative processes in industrial production, e.g. for alternative packaging options and it was further developed and applied to an increasing variety of product types in the following decades. One of the indicators included in LCAs, the so called *global warming potential* (GWP), depicts GHGEs occurring throughout a product's life cycle. Similarly, a *carbon footprint* (CF) – following a rather broad definition, which includes CH_4 and N_2O besides CO_2 (e.g. BSI, 2008; WRI/WBCSD, 2009) – also quantifies the GHGEs occurring throughout a product's life cycle or at least along its supply chain.

One main goal of this doctoral thesis was to suggest a basis for a proper inclusion of all relevant GHGE into product assessments. Especially emissions from imported inputs, such as feedstuffs, which were mostly not covered in previous studies, shall be considered. Earlier studies on GHGE from agricultural production were often restricted to processes occurring on farm. Emissions which occurred before the farm gate were frequently excluded, as were GHGEs from abroad if estimations were based on national GHG-inventories. Therefore the majority of previous studies did not really fulfil recommendations for LCAs and CFs unless they cover all relevant fluxes of GHGEs throughout the life cycle or at least the whole supply chain. Therefore this thesis aims at covering sources for GHGE which were ignored in previous studies.

The thesis consists of an introduction, three scientific articles, a general discussion and conclusion section, which is meant to provide a synopsis and perspectives for the fields covered.

The first article was published in the Journal of Renewable Agriculture and Food Systems (no. 25; pp 316-329; doi: 10.1017/S1742170510000025). It provides an overview of GHGE from dairy production with a focus on (i) the sound modelling of typical Austrian dairy production systems, (ii) the further development of methods to assess GHGEs along the whole production chain and (iii) on derivation of mitigation strategies for Austrian milk production systems.

As imported dietary components rich in protein (i.e. especially soybean meal) were identified as one key source of GHGE, the second publication focussed on the substitution of GHGE-loaded feedstuffs as one important mitigation option. This article was published in the Journal of the Science of Food and Agriculture (no. 91; pp 1118–1127; doi: 10.1002/jsfa.4293).

The issue of GHGEs which result from a change in land use pattern (e.g. clearance of forests or ploughing of grasslands) is of particular relevance for imported protein feedstuffs, but was not considered in most previous studies. Therefore the issue of GHGEs which originate from land conversion and the basic methodical aspects connected to a sound emission assessment were addressed in the third publication, which has been submitted to the Journal Greenhouse Gas Measurement & Management (submission in November 2011).

2 Journal contributions

2.1 Greenhouse gas emissions from selected Austrian dairy production systems: model calculations considering the effects of land use change

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Greenhouse gas emissions from selected Austrian dairy production systems—model calculations considering the effects of land use change

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Abstract

The aim of this study was to analyze various Austrian dairy production systems (PS) concerning their greenhouse gas emissions (GHGE) in a life-cycle chain, including effects of land-use change (LUC). Models of eight PS that differ, on the one hand, in their regional location (alpine, uplands and lowlands) and, on the other hand, in their production method (conventional versus organic, including traditional and recently emerging pasture-based dairy farming) were designed.

In general, the GHGE-reducing effect of a higher milk yield per cow and year in conventional dairy farming cannot compensate for the advantages of organic dairy production which requires lower inputs. This is shown both for GHGE per kg of milk and GHGE per ha and year of farmland. Especially when (imported) concentrates were fed, which had been grown on former forests or grassland, e.g. soybean meal and rapeseed cake, GHGE of conventional dairy farming rose due to the effects of LUC.

GHGE per kg milk varied from 0.90 to 1.17 kg CO_2-eq for conventional PS, while organic PS on average emitted 11% less greenhouse gases (GHGs), the values ranging from 0.81 to 1.02 CO_2-eq per kg milk. Within each production method, PS with a higher milk output generally showed better results for GHGE per kg of milk produced than PS with a lower milk output. Nevertheless the latter showed clearly better results for GHGE per ha of land used, ranging from 5.2 to 7.6 Mg CO_2-eq per ha and year for conventional PS and from 4.2 to 6.2 Mg CO_2-eq per ha and year for organic PS. The results of this study emphasize the importance of a complete life-cycle assessment in the evaluation of impacts that dairy PS have on the climate.

Key words: dairy cow, milk, greenhouse gas emissions, land-use change, mitigation

Introduction

Agriculture, especially animal husbandry, causes considerable greenhouse gas emissions (GHGE). In the EU-15, agriculture accounted for approximately 10% of total GHGE in 2000[1]. On the one hand, cattle and other ruminants emit relatively large quantities of greenhouse gases (GHGs), particularly methane from enteric fermentation. On the other hand, a large percentage of the agriculturally utilized land in Austria is located in mountainous areas and uplands[2]. As in other regions that are dominated by grassland, cattle and other ruminants are an essential element of regional agricultural food production. Cessation of (livestock) farming in these regions would therefore have tremendous socio-economic and high ecological costs[2,3].

Grasslands, pasture and (tropical) forests are vegetations with a high environmental value, with high biodiversity and carbon storage potential[4,5]. Land-use change (LUC), especially in combination with forest clearing in the tropics, is assumed to cause up to one-quarter of anthropogenic CO_2-emissions[5]. However, the relative contribution of LUC to GHGE from traditional grassland-based dairy production through imported concentrates has not yet been studied[6]. Another source that is sometimes not considered in estimations of GHGE is the emissions occurring during the rearing period of heifers, on the one hand; on the other hand, beef from calves and cull cows are important by-products from dairy production which result in a relative reduction of GHGE accountable to milk production, but are nevertheless not always considered in respective calculations (e.g. Löthe et al.[7]).

Given the regional importance of agriculture in general, and especially of dairy production in alpine regions of Europe, local traditional production systems (PS) must be further developed, including a reduction of their GHGE. Nevertheless, for these PS mitigation options have not yet been extensively analyzed in the literature as most studies have covered more intensive lowland PS which were characterized by greater livestock density and higher quantities of purchased production factors, such as bought-in feed or chemicals (e.g. Thomassen et al.[8], Williams et al.[9] and Cederberg and Mattson[10]).

Therefore the goals of this study were to estimate the level of GHGE for selected milk PS in Austria, taking a great number of sources for GHGE into account, to analyze relevant influencing factors and to identify options for their reduction.

Material and Methods

Models for different PS were built, using MS Excel for calculation and taking into account emissions of methane (CH_4) and nitrous oxide (N_2O) from enteric fermentation and from manure management, as well as of CH_4, N_2O and carbon dioxide (CO_2) from soil, from the use of fuels and other energy sources and from production and application of mineral fertilizers and pesticides. Total emissions per cow and year, per kg milk and per ha of farmland used were calculated by adding up the emissions of CH_4, N_2O and CO_2 as CO_2-equivalents (CO_2-eq). Conversion factors used to calculate the global warming potential are 23 kg CO_2-eq for 1 kg methane and 296 kg CO_2-eq for 1 kg N_2O (100-year-horizon)[11].

Modeled dairy PS

Models for Austrian dairy PS were built on farm-level for alpine regions, uplands and lowlands, each one for organically and conventionally managed PS. PS a (alpine) represents a traditional alpine farm at more than 800 m asl and steeply sloping grassland. PS U (uplands) represents a dairy farm in a region of transition from alpine to lowlands with about 20% of arable land and 80% of permanent grassland, while PS L (lowlands) is located in the lowlands, farming on permanent grassland and arable land at a ratio of about 50 : 50. Generally, the intensity of production (i.e., among others, stocking rate and milk yield per cow) increases within these PS (PS A < PS U < PS L), while PS UP (uplands, pasture) represents a low-input, pasture-based production system with a relatively high stocking rate. Herein, the appendices '$_{org}$' and '$_{con}$' are used to further differentiate between organic and conventional PS, respectively. These PS may not necessarily represent the average Austrian dairy farm, but rather represent a wide spectrum of different conditions for dairy production. The great variability in farming conditions in Austrian dairy production are due to the geographic heterogeneity, to the originally small farm size and the different development during the past three decades. According to an analysis, which was based on a comprehensive Austrian agricultural statistical database (Invekos)[12], about 14% of Austrian dairy farms could be assigned to the PS A_{con}, A_{org}, UP_{con} and UP_{org} (between 4500 and 6000 kg milk quota per cow and year in an alpine or upland region), with a relatively high share (26%) of organic farms. Within these 14%, pasture-based PS contribute an estimated 15%. Although currently only relatively few farms exist which are strictly following a pasture-based low-input approach with the highest possible proportion of pasture (up to 60%; Table 1) and seasonal calving, this concept of dairy production is extensively discussed as a future strategy of milk production in permanent grassland regions[13]. PS UP_{con} and UP_{org} were defined to represent farms following such a strategy. PS U_{con} and U_{org} represent another 8% of Austrian dairy farms (i.e., with a share of grassland between 65 and 85%), with 16% of farms being organically managed. Furthermore, about 4% of Austrian dairy farms are represented by PS L_{con} and L_{org} with a milk quota of 6500–8000 kg milk per cow (among them about 9% of organic farms).

Due to differences in management (e.g., number of harvests, dietary proportion and type of bought-in and home-grown feed), milk yield and usage of fuel, mineral fertilizers and pesticides, among other factors, differ between the eight PS. Key characteristics for these PS are shown in Tables 1, 2 and 3.

Crop and grassland yields, feeding value. Values for crop yields were derived from Austrian statistical databases[14], differentiating between conventional and organic production. Yields for grassland were taken from Buchgraber and Gindl[15] and an agricultural national database[16] and were equal for conventional and organic production but were adjusted to the altitude. The main reasons for the assumption of equal forage yields were similar amounts of manure applied, lower gaseous N-losses in organic housing due to higher proportions of pasture and straw-based manure systems (20 and 30% gaseous N-losses from NH_3 and NO_x emissions for pasture and storage of solid manure, respectively, as compared to 40% N-losses for slurry systems; IPCC[17], Tables 10.22 and 11.3) and a higher share of legumes in organic grassland and hence higher biological N-fixation (e.g. Rahmann and Böhm[18]). Information on nutrient contents of crops and forage were derived from DLG feed tables[19], Buchgraber

Table 1. Emissions from manure management, direct and indirect soil emissions for the eight PS.

Trait	PS A_{con}	PS A_{org}	PS UP_{con}	PS UP_{org}	PS U_{con}	PS U_{org}	PS L_{con}	PS L_{org}
Percentage of manure excreted in straw-based systems (%)	43	40	24	24	60	51	60	51
Percentage of manure excreted in slurry-based systems (%)	29	26	16	16	40	33	40	33
Percentage of manure excreted on pasture (%)	28	34	60	60	0	16	0	16
N_2O—manure (kg cow^{-1} year^{-1})	1.2091	1.1280	0.6688	0.6791	1.8388	1.5220	1.9513	1.5688
CH_4—manure (kg cow^{-1} year^{-1})	25.49	23.30	15.67	15.43	37.13	30.32	39.26	31.26
CO_2-equivalents—manure (kg cow^{-1} year^{-1})	944.2	869.8	558.4	555.9	1398.3	1147.9	1480.6	1183.3
N_2O-emitting soil-N (kg cow^{-1} year^{-1})	114.5	110.6	70.1	70.1	112.5	102.4	95.3	98.8
Direct N_2O soil emissions (kg cow^{-1} year^{-1})	2.16	2.16	1.61	1.61	1.93	1.73	1.50	1.67
Indirect N_2O soil emissions from deposition and leaching (kg cow^{-1} year^{-1})	0.62	0.62	0.59	0.59	0.70	0.66	0.77	0.67

et al.[20] and Resch et al.[21]. The management and production factors for the different PS were defined based on Austrian agricultural statistical data[14,16,22,23], and are described in Tables 2 and 3.

Forage quality depended, among others, on altitude, cutting frequency and time of harvest or grazing, ranging from 5.35 to 6.15 MJ NE_L ('net energy lactation'[24,25]) per kg of dry matter (DM)[15,21].

Feeding regimen and milk yields. The relative proportion of pasture in the total forage was assumed to decrease from PS UP via PS A to PS U; in PS L_{con}, cows were not grazed anymore. The dietary percentage of hay also decreased in the same order, whereas the percentage of silage increased. In general, organic PS fed more hay and used more grazing than conventional PS, as it is shown in Table 2. In the most intensive PS L_{con}, about 40% of grass-clover silage were replaced by maize silage. Forage harvest-losses were related to the type of forage fed: pasture 25%, indoor grass feeding 5%, grass silage 20% and hay 30%, on a DM basis[16].

The annual average percentage of concentrate in the diets was assumed to be between 13% of total feed intake in PS UP (conventional and organic) and 24% in PS U_{con} and PS L_{con}. In the pasture-based PS UP_{con} and PS UP_{org}, the concentrate only consisted of grains and mineral premix. Organic concentrate consisted of barley, wheat, faba beans, peas and mineral premix for PS A_{org}, PS U_{org} and PS L_{org}. Conventional concentrate contained barley, wheat, corn, rapeseed cake, extracted soybean meal, faba beans and mineral premix for PS A_{con}, PS U_{con} and PS L_{con}. Forty-five percent of the rapeseed for oil milling (from which rapeseed cake originates as a by-product) was assumed to be imported from abroad (mainly from European countries such as Hungary and Slovakia), the rest was produced in Austria[26]. It was assumed that the production and extraction of soybeans took place mainly in South America and Germany, respectively[27]. While PS A and PS UP had to buy-in all the concentrates, PS U_{con}, PS U_{org}, PS L_{con} and PS L_{org} bought-in 56, 58, 53 and 24% of total concentrates, respectively.

Milk yields ranged from 5500 kg per lactation (organic and conventional PS UP and PS A) to 8000 kg per year (PS L_{con}), with an estimated average lifetime performance of 23,650 kg milk, as was recorded for Austria's main breed, Simmental[28].

Livestock density per hectare (stocking rate) was related to the PS as well as the feeding strategy and was between 1.0 and 1.5 livestock-units of dairy cows per hectare. Internal farmland was assumed to be between 0.67 and 1.0 ha per cow, but due to the demand on land for the production of bought-in feed, total farmland required ranged from 0.84 to 1.23 ha per cow (Table 3). Although cash crops may be produced, particularly in PS U and PS L, only land for cattle feed production was considered herein.

Sources of emissions

Enteric fermentation. Unlike CO_2 emissions from livestock, which are assumed to be zero due to photosynthesis of plants, emissions of CH_4 have to be considered according to IPCC[11]. CH_4 emissions from enteric fermentation were estimated using an equation established by Kirchgeßner et al.[29]:

$$CH_4 = 63 + 79\,CF + 10\,NfE + 26\,CP - 212\,EE,$$

where 'CH_4' describes the enteric methane emissions (in g), 'CF' is dietary crude fiber (in kg), 'NfE' is the dietary easily soluble carbohydrates (N-free extracts; in kg), 'CP' is the dietary crude protein (in kg) and 'EE' is the ether extracts (in kg). Table 4 shows some characteristic traits for the rations (Table 2) fed in the eight PS.

Table 2. Characteristics of feed production, feed quality and dietary composition for the eight PS.

Characteristic	**PS A_{con}**	**PS A_{org}**	**PS UP_{con}**	**PS UP_{org}**	**PS U_{con}**	**PS U_{org}**	**PS L_{con}**	**PS L_{org}**
Average number of cuts or frequency of pasturage for permanent grassland and clover ley (*n*)	2	2	4	4	3	3	4.5	4.5
Gross yields of permanent grassland/clover ley ($Mg\,DM\,ha^{-1}\,year^{-1}$)	6.0	6.0	9.0	9.0	8.0/8.5	8.0/8.5	10.0/11.0	10.0/10.5
Average energy density of forage[1] ($MJ\,NE_L\,kg^{-1}\,DM$)	5.40	5.35	5.90	5.90	5.60	5.60	6.15	5.80
Energy density of concentrate ($MJ\,NE_L\,kg^{-1}\,DM$)	8.05	8.05	8.00	8.00	8.05	8.05	8.05	8.05
Overall percentage (type[2]) of forage per kg diet-DM (%)	83 (grass silage, hay, pasture)	81 (hay, pasture, grass silage)	89 (pasture, hay, grass silage)	89 (pasture, hay, grass silage)	76 (grass-clover silage, pasture, hay, fresh grass)	83 (grass-clover silage, pasture, hay, fresh grass)	76 (grass-maize-clover silage)	81 (grass-clover silage, pasture, hay, fresh grass)
Overall percentage of concentrate per kg diet-DM (%)	17	19	11	11	24	17	24	19
Percentage of bought-in feed per cow (% DM)	17	19	11	11	14	10	13	5

[1] Including clover silage and maize silage in uplands and lowlands PS if existing.
[2] Feedstuffs ranked according to their dietary percentage.

Energy consumption. The energy directly used on farm for dairy production, as well as the fuels and electric energy that were consumed during the production of mineral fertilizers and pesticides, were considered herein. The greatest share of the electric energy needed on dairy farms is used to cool milk and to produce and supply (concentrate) feed. Therefore, the amount of energy used was related to the annual milk yield, also considering that more feed was required and that higher-mechanized (energy-consuming) housing systems coincided with higher milk yields. A value of 0.05 kWh per kg milk was assumed[30]. All calculations in this model were done per cow and did not account for differences in farm size. Emissions were estimated to be 0.453 kg CO_2-eq per kWh on average[31]. The amount of fuel used for cultivating the fields was estimated using standard values from a national database[32], resulting emissions were calculated according to Fehrenbach et al.[4]. Additionally, the energy needed in transporting externally produced feedstuffs was also taken into account according to Wilting et al.[33]. Mineral fertilizers and pesticides must not be used in the organic PS and were also not used on the grassland of the conventional PS, PS A_{con} and PS UP_{con}. However, a proportionate input of these factors was accounted for the concentrates imported into the conventional PS.

Emissions occurring during the production of mineral fertilizers and pesticides were derived from Patyk and Reinhardt[34] and Biskupek et al.[35]. Table 5 shows energy-related emission factors used herein and the references from which the data were derived. PS A_{con} is used as an example to demonstrate how the emissions attributed to these sources were calculated (Table 6).

Construction of machinery and buildings were not included as sources of emissions in the model calculations, as they were expected to be equal for all PS.

Manure management. The manure systems were assumed to represent the situation in Austria: 60.7 and 59.7% of the organic and conventional dairy cows, respectively, are housed in straw-based systems[23]. The remaining systems are slurry-based. Therefore, differences occur between the eight PS, according to the proportion of time spent on pasture; according to Amon et al.[22,23], the amount of manure per cow, its organic DM and nitrogen (N) contents, which are in turn related to milk yield and feed intake. Representative data were derived from Gruber and Steinwidder[36], the amounts of manure were calculated to be between 19.4 (for both PS A and PS UP) and 22.4 (for PS L_{con}) Mg per cow and year. The content of volatile solids excreted daily (DM) was calculated to be between 4.0 and 4.6 kg per cow, based on Schechtner[37]. The quantity of nitrogen excreted was estimated to vary from 86.2 to 100.6 kg per cow and year if a moderate N-content of feed is assumed[38]. Based on these values, the amounts of CH_4 and N_2O emitted were estimated according to IPCC[17] (tier 2; equations 10.23 and 10.25) and are given in Table 1, using methane conversion factors of 0.3, 0.04 and 0.015 for slurry, farmyard

Table 3. Key characteristics of the eight PS.

Characteristic	**PS A_{con}**	**PS A_{org}**	**PS UP_{con}**	**PS UP_{org}**	**PS U_{con}**	**PS U_{org}**	**PS L_{con}**	**PS L_{org}**
Regional location, production method	Alpine, conventional	Alpine, organic	Uplands pasture-based, conventional	Uplands pasture-based, organic	Uplands, conventional	Uplands, organic	Lowlands, conventional	Lowlands, organic
Stocking density (cow-LU ha^{-1})	1.0	1.0	1.4	1.4	1.2	1.1	1.5	1.2
Internal farmland required per cow[1] (ha)	1.0	1.0	0.71	0.71	0.83	0.91	0.67	0.83
Total farmland required per cow[2] (ha)	1.23	1.34	0.84	0.88	1.11	1.10	0.94	0.92
Permanent grassland, proportion of agricultural land (%)	100	100	100	100	80	80	50	56
Arable land, proportion of agricultural land (%)	0	0	0	0	20	20	50	44
Crop rotation on arable land and percentage of the crops (%)	–	–	–	–	Clover ley (20), wheat (25), barley (55)	Clover ley (25), barley (34), faba beans (25), wheat (16)	Clover ley (20), maize (35), barley (30), wheat (15)	Clover ley (25), barley (34), faba beans (25), wheat (16)
Annual milk yield per cow (kg)	5500	5500	5500	5500	7000	6500	8000	7000
Milk yield per ha of total farmland (kg)	4475	4103	6576	6223	6304	5913	8542	7606
Total farmland required per 1000 kg milk (ha)	0.223	0.244	0.152	0.161	0.159	0.169	0.117	0.131

[1] For the production of homegrown feedstuffs.
[2] For the production of homegrown plus bought-in feedstuffs.

Table 4. Nutrient intake and their relative dietary proportions for the eight PS.

Nutrient	PS A_{con}	PS A_{org}	PS UP_{con}	PS UP_{org}	PS U_{con}	PS U_{org}	PS L_{con}	PS L_{org}
CF (kg per day; [%])	3.43 [24]	3.47 [24]	3.44 [25]	3.44 [25]	3.27 [22]	3.51 [23]	2.98 [18]	3.21 [22]
NfE (kg per day; [%])	7.09 [50]	7.25 [51]	6.66 [48]	6.66 [48]	7.61 [50]	7.47 [49]	9.08 [56]	7.12 [50]
CP (kg per day; [%])	1.98 [14]	1.91 [13]	2.03 [15]	2.03 [15]	2.34 [15]	2.26 [15]	2.31 [14]	2.18 [15]
EE (kg per day; [%])	0.41 [3]	0.36 [3]	0.34 [2]	0.34 [2]	0.48 [3]	0.44 [3]	0.52 [3]	0.42 [3]
Ash (kg per day; [%])	1.34 [9]	1.29 [9]	1.41 [10]	1.41 [10]	1.45 [10]	1.52 [10]	1.25 [8]	1.41 [10]

CF, dietary crude fiber; NfE, dietary easily soluble carbohydrates; CP, dietary crude protein; EE, ether extracts.

Table 5. Overview on used emission factors.

Process	Unit	Emission factor	Reference
Fuels—emissions from supply chain and consumption	kg CO_2-eq l^{-1} diesel	3.2066	Fehrenbach et al.[4]
Electric energy—emissions from supply chain	kg CO_2-eq kWh^{-1}	0.453	Ecoinvent[31]
Production mineral fertilizer—nitrogen (N)	kg CO_2-eq kg^{-1} N-mineral	7.470	Patyk and Reinhardt[34]
Production mineral fertilizer—phosphorus (P_2O_5)	kg CO_2-eq kg^{-1} P_2O_5-mineral	1.176	Patyk and Reinhardt[34]
Production mineral fertilizer—potassium (K_2O)	kg CO_2-eq kg^{-1} K_2O-mineral	0.664	Patyk and Reinhardt[34]
Production pesticides	kg CO_2-eq kg^{-1} pesticide	5.369	Biskupek et al.[35]

manure and pasture, respectively. Emission factors used for calculation of N_2O were 0.02 and 0.001 for farmyard manure and slurry, respectively[17].

Soil N_2O. Direct N_2O emissions were calculated based on the amount of nitrogen introduced into the soil (IPCC-tier 2, equation 11.2)[17]. Amounts of N from mineral fertilizers, mineralization, manure and crop residues were multiplied by the default emission factors of 0.01 kg N_2O-N per kg of N applied; a factor of 0.02 kg N_2O-N was used for each kg of N excreted by cows on pasture. Indirect soil emissions from deposited nitrogen and leaching were estimated according to IPCC[17] (default values in Tables 10.22 and 11.3). In Table 1, amounts of N and related N_2O-emissions are shown for the different PS. N_2O emitted during cultivation of bought-in concentrates was included, because of its relevance for the N-balance of the total dairy supply chain. Due to a low demand on farmland and less grazing, soil N_2O-emissions per cow seem to be lower for more intensive PS. On the contrary, emissions from manure are higher for more intensive systems with little or no grazing (Table 1).

LUC and changes in soil organic carbon stocks. LUC for soybean production was calculated according to statistics on imports (98% of imported soybean meal originated from South America[27]) and based on estimates for the conversion of savannah-type vegetation into soybean fields[4]. The emissions from LUC, which may be even higher in the case of rainforest clearance, were calculated depending on loss of carbon from the soil and aboveground biomass and were allocated to soybean oil and extracted soybean meal based on their caloric values.

As supported by data from the European Environment Agency (EEA)[39], LUC from grassland to arable land could be neglected for Austria, but had to be considered for imports of rapeseed from Eastern and Central Europe. An equivalent of 53% of LUC-related GHGE was estimated for the CO_2-eq value of rapeseed cake by Fehrenbach et al.[4], which was used as feedstuff for dairy cattle, because 45% of the rapeseed further processed in Austria was imported mainly from Eastern and Central Europe in 2004[26]. Consequently, LUC-related GHGE of 5.41 and 0.40 kg CO_2-eq were calculated for soybean meal and rapeseed cake, respectively, in addition to emissions from cultivating, transport and processing.

CO_2 sequestrated into soil or released from soil organic carbon stocks was calculated according to Küstermann et al.[40] for Bavaria, Germany, where on-site conditions can be expected to be very similar to those in Austria. CO_2-emissions and -sequestration occur for conventionally (+202 kg ha^{-1} a^{-1}) and organically managed soils (−400 kg ha^{-1} a^{-1}), respectively, due to differences in crop rotation and manure management. An even higher sequestration rate of 575 kg CO_2 ha^{-1} a^{-1} was observed in a previous long-term study in Switzerland[41], where on-site conditions could also be expected to be similar to those in Austria.

It was estimated that 1 kg of conventional concentrate caused emissions of 0.05 kg CO_2-eq from soil organic carbon changes, based on +202 kg CO_2-eq ha^{-1} a^{-1} and an average grain yield of 4000 kg ha^{-1} a^{-1}. One kilogram of an organic concentrate was expected to be related to a sequestration of 0.111 kg CO_2-eq (−400 kg CO_2-eq ha^{-1} a^{-1} and an average grain yield of 3600 kg ha^{-1} a^{-1}). Due to its long history of relatively constant management, it was assumed that Austrian alpine grassland is at an equilibrium state and that its soils did not emit or sequestrate further CO_2[42].

Rearing phase and beef as a by-product. The rearing phase of dairy cows prior to first calving has to be considered as an important source of GHGE, together with

Table 6. Consumption of energy and inputs per cow and year: PS A_{con} as an example.

Sources	Demand for fuels (litres)	Demand for additional mineral fertilizer[1] (kg $N/P_2O_5/K_2O$)	Demand for pesticides (kg)	Demand for electric energy (kWh)	Emissions from fuels (kg CO_2-eq)	Emissions from production of mineral fertilizer (kg CO_2-eq)	Emissions from production of pesticides (kg CO_2-eq)	Emissions from electric energy (kg CO_2-eq)
Reference	ÖKL[32], Wilting et al.[33]	BMLFUW[16]	Biskupek et al.[35]	Öhlinger et al.[30]	Fehrenbach et al.[4]	Patyk and Reinhardt[34]	Biskupek et al.[35]	Ecoinvent[31]
Forage (homegrown)	58.2	9.8/0/85.0	0	–	186.6	129.6	0	–
Concentrate (bought-in)	17.9	18.6/7.1/6.1	0.9	–	57.4	149.6	4.8	–
Transports	13.3	–	–	–	42.6	–	–	–
Housing	–	–	–	275	–	–	–	124.6

[1] Taking into account the amount of nutrients in the manure.

beef as the by-product from dairy production having a related mitigating effect for GHGE per kg of produced milk.

In the model calculations, the GHGE during the rearing phase were calculated for each PS based on the average emissions per MJ NE_L consumed during the rearing phase. In the same way, emissions for a growing–fattening heifer were calculated as a standard of comparison for each PS, based on the emissions per MJ NE_L. The mitigating effect of beef as a by-product from cull cows and newborn calves (50% bodyweight estimated as carcass) was calculated, using growing–fattening heifer as a model for beef from each PS. By considering all sources of GHGE except electric energy for cooling milk, 1 MJ NE_L was burdened with 0.15 kg CO_2-eq (PS L_{org}) to 0.19 kg (PS A_{con}). The total requirement for energy during a dairy cows' rearing phase was assumed to be about 31,000 MJ NE_L with an age at first calving of 28 months[28,43]. As a consequence, emissions during the rearing phase were calculated to vary from 4.7 for PS L_{org} to 5.9 Mg CO_2-eq per cow in PS A_{con}. Emissions during the growing–fattening of a heifer (about 20,000 MJ NE_L required for 600 kg final body weight) were estimated to vary from 3.0 to 3.8 Mg CO_2-eq per head; the mitigating effect of beef from slaughter cows and newborn calves was therefore calculated to be between 3.1 and 3.9 Mg CO_2-eq per cow.

Results and Discussion

GHGE from dairy PS

Emissions related to input factors and milk yield. Total emissions per cow (related to a lifetime milk yield of 23,650 kg) are presented in Table 7. The majority of GHGE evolved from enteric fermentation (40–62%), while the use of fuels and energy (in total 5–9%) as well as production of external inputs such as mineral fertilizers and pesticides (up to 7%) contributed relatively little. In Figure 1, GHGE from soil (N_2O), from fuels, fertilizers and pesticides used, were aggregated to GHGE accountable to forage and concentrate supply. As the same sources—except electric energy for cooling milk—contribute to emissions during the rearing period and to the mitigation of emissions indirectly caused by the by-product beef, total GHGE from enteric fermentation are actually higher than presented in Figure 1 and Table 7. For example, total GHGE from enteric fermentation are 53% of total emissions (i.e., 3326 kg CO_2-eq per year) as compared to 49% (i.e., 3100 kg CO_2-eq annually) which are attributed to one productive year of a dairy cow.

In previous calculations, which focused on the relation between GHGE and milk yield, emissions which originated from the rearing phase were not always taken into account (e.g., Löthe et al.[7]). Although milk yield remains to be an essential factor, the rearing phase is equally important, as a heifer needs nearly the same amount of energy for growth and maintenance as a cow during one lactation. Additionally, the production of beef as a by-product of dairy

Table 7. Emissions of GHGs ($kg\,CO_{2\,eq}\,cow^{-1}\,year^{-1}$; [%]) from different sources for the eight PS, including the source 'rearing phase' and beef as a by-product.

Source of emission	PS A_{con}	PS A_{org}	PS UP_{con}	PS UP_{org}	PS U_{con}	PS U_{org}	PS L_{con}	PS L_{org}
Rearing phase—proportional emissions per lactation	1359 [21]	1167 [21]	1210 [21]	1198 [23]	1686 [23]	1314 [22]	1679 [23]	1359 [24]
Enteric fermentation	3100 [48]	3215 [57]	3209 [57]	3209 [62]	2995 [42]	3205 [54]	2841 [40]	2988 [52]
Manure	944 [15]	870 [16]	558 [10]	556 [11]	1398 [19]	1148 [19]	1481 [21]	1183 [21]
Forage—fuels	187 [3]	179 [3]	133 [2]	133 [3]	189 [3]	199 [3]	165 [2]	193 [3]
Forage—inputs	130 [2]	0 [0]	264 [5]	0 [0]	167 [2]	0 [0]	140 [2]	0 [0]
Forage—direct N_2O	547 [8]	561 [10]	564 [10]	564 [11]	409 [6]	426 [7]	223 [3]	281 [5]
Concentrates—fuels	100 [2]	106 [2]	37 [1]	58 [1]	161 [2]	125 [2]	226 [3]	177 [3]
Concentrates—inputs	154 [2]	0 [0]	104 [2]	0 [0]	125 [2]	0 [0]	166 [2]	0 [0]
Concentrates—direct N_2O	86 [1]	78 [1]	58 [1]	58 [1]	150 [2]	122 [2]	257 [4]	182 [3]
Indirect soil emissions—N_2O	183 [3]	179 [3]	176 [3]	176 [3]	206 [3]	196 [3]	228 [3]	199 [3]
Changes in soil organic carbon	48 [1]	− 113 [− 2]	37 [1]	− 81 [− 2]	73 [1]	− 109 [− 2]	77 [1]	− 121 [− 2]
Production of energy in housing	125 [2]	125 [2]	125 [2]	125 [2]	159 [2]	147 [2]	181 [3]	159 [3]
By-product beef—proportional emissions per lactation	− 895 [− 14]	− 772 [− 14]	− 801 [− 14]	− 793 [− 15]	− 1116 [− 16]	− 870 [− 15]	− 1112 [− 15]	− 899 [− 16]
LUC	389 [6]	0 [0]	0 [0]	0 [0]	589 [8]	0 [0]	628 [9]	0 [0]
Total GHGE	6452 [100]	5594 [100]	5675 [100]	5203 [100]	7190 [100]	5902 [100]	7181 [100]	5699 [100]
Total GHGE related to milk yield ($kg\,CO_{2\,eq}\,kg^{-1}$ milk)	1.173	1.017	1.032	0.946	1.027	0.908	0.898	0.814
Total GHGE related to total farmland ($kg\,CO_{2\,eq}\,ha^{-1}$)	5246	4175	6756	5913	6477	5365	7639	6195

production, lifetime performance and the number of lactations that a cow lasts, are important factors as well. Following Fürst[44], conventional and organic Austrian dairy cows are expected to produce about the same amount of milk in their lifetime, but with the latter being different.

Generally, PS with a higher output of milk showed higher GHGE per cow and year but were superior over PS with a lower output if emissions were expressed per kg of milk. GHGE per kg of conventional and organic milk were between 0.90 and 1.17 kg CO_2-eq per kg of milk and between 0.81 and 1.02 kg CO_2-eq per kg of milk, respectively (Table 7, Fig. 1). These numbers illustrate that differences may be smaller between production methods (i.e. conventional versus organic) than between regions: on average, organic PS showed about 11% lower GHGE per kg milk than comparable conventional PS, while the relative difference between the lowest emissions from the PS L and the highest emissions from the PS A amounted to 22%. Generally, the higher the dietary energy and nutrient density, the higher the milk yield, which results in lower GHGE per kg of milk due to reduced enteric fermentation and performance-related degression.

As the nutritional value of forages and concentrates is routinely characterized by proximate analysis in Austria, enteric fermentation was calculated according to the regression equation published by Kirchgeßner et al.[29]. In order to check these estimates, a regression equation derived by Hindrichsen et al.[45] was used. It was found that estimates resulting from the equation by Kirchgeßner et al. were on average 8% higher than those resulting from the equation of Hindrichsen et al., with the difference being greater for rations that are rich in crude fiber and being lower for rations rich in soluble carbohydrates.

The role of LUC. Another highly relevant source of GHGE has to be taken into account whenever conventional PS utilize soybean meal originating from South America: the production of soybeans is linked to an LUC from former savannah-type vegetation into arable land. In contrast to most previous calculations of GHGE and life-cycle assessments which did not incorporate LUC in their calculations (e.g., Lehuger et al.[46]), LUC was taken into account in this study (Table 7, Figs. 1 and 2) due to its high relevance on the global scale[4,5,11]. Estimates for GHGE from LUC vary considerably, e.g., about 5 kg CO_2-eq per kg of Argentinean soybean meal[47] as compared to more than 10 kg CO_2-eq per kg of Brazilian soybean meal. The latter occurs if soybean is cultivated on deforested land (calculated from Renewable Fuels Agency[48]). Herein, a change from extensive grassland (savannah) to arable land was assumed[4], but without deforestation. Since 2000, the area for soybean cultivation expanded predominantly on deforested land[49] and therefore emissions from LUC may actually be even higher than calculated by both Fehrenbach et al.[4] and within this study. Nevertheless, even with relatively low estimates for GHGE from LUC, emissions per kg milk will be lower for PS which do not import soybean meal into their

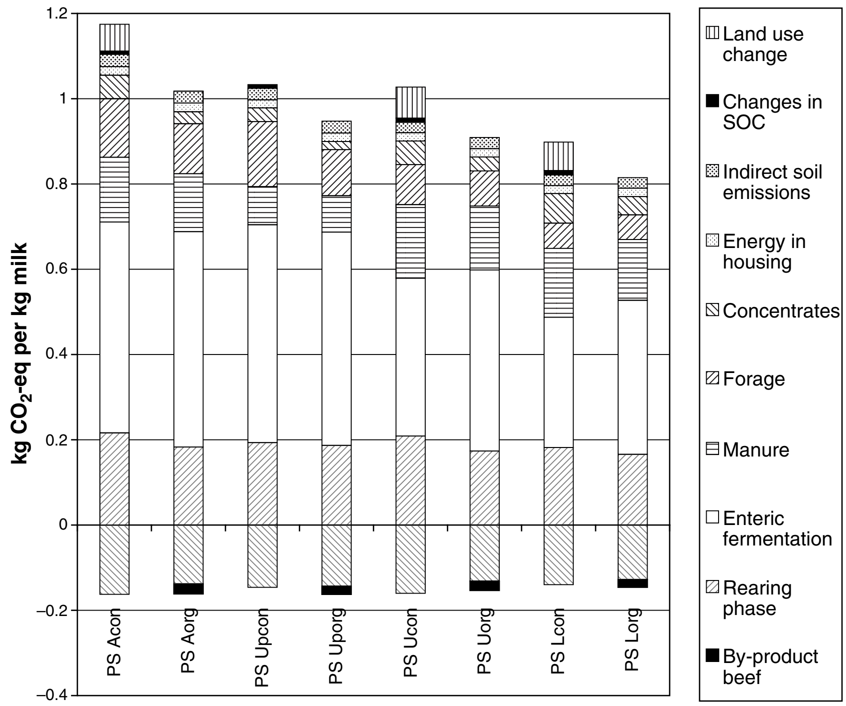

Figure 1. GHGE (kg CO_2-eq) per kg milk for the eight PS.

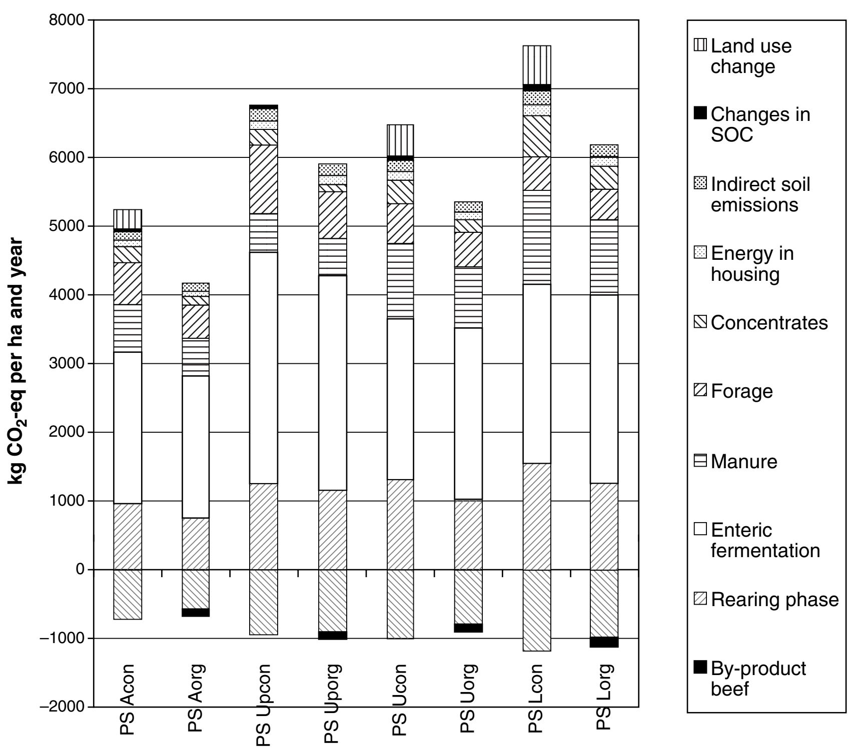

Figure 2. GHGE (Mg CO_2-eq) per ha and year of total farmland for the eight PS, including the source, rearing phase, and beef as a by-product.

system, but utilize homegrown or locally produced protein sources such as grain legumes or oilseed cakes. This contributes to the average difference in GHGE of 11% between conventional and organic PS studied herein. About 8% of total GHGE in conventional farming (average over all conventional PS excluding PS UP_{con}) result from LUC, mainly (93%) for extracted soybean meal originating from South America.

Emissions and regional location of dairy production. The difference in GHGE between alpine and lowlands PS with both conventional and organic management is mainly due to higher milk yields in the lowlands as well as a decreased enteric fermentation as a consequence of increased dietary energy density. Pasture-based PS (UP) emit relatively low amounts of GHGE per kg milk, despite their comparably low milk yield of 5500 kg per cow and year. This is caused by several factors, including reduced emissions from manure in housing (with cows being on pasture for 60% of the time) and the use of low amounts of concentrates. On the contrary, GHGE from enteric fermentation are slightly higher as compared to PS that use less hay but more silage. Pasture-based systems also have higher GHGE from forage because of a doubled rate of N_2O from N excreted during grazing[17], as compared to N spread as manure. Both the conventional and organic PS UP show a high productivity per ha of farmland required and low GHGE, which are similar to those of PS U.

Only very few studies are available about GHGE from dairy PS located in alpine or nearby regions (e.g. Weiske et al.[50] and Olesen et al.[51]). In contrast to the results presented herein, Weiske et al., using a model based on Olesen et al., reported generally higher GHGE per kg milk of between 1.2 and 2.0 kg CO_2-eq although LUC was not included in their calculations. The differences may be due to the lower milk yield and a lack of differentiation between organic and conventional PS in milk yield as assumed by Weiske et al., a greater number of input factors considered (e.g., the production of mineral premix and seeds), but also disregarding the emission-mitigating effects of by-products.

Great differences occur between PS when GHGE are related to the area of farmland used (Fig. 2). From this perspective, low-input systems show clearly better results. Organic PS need more area per cow due to lower yields especially from arable land, and at the same time show less GHGE per cow. As presented in Figure 2, GHGE per hectare of total farmland vary from 5.2 to 7.6 Mg CO_2-eq and from 4.2 to 6.2 Mg CO_2-eq for conventional and organic PS, respectively. Olesen et al.[51] reported comparable results of 8.7 Mg CO_2-eq for conventional PS and 6.0 Mg CO_2-eq for organic PS calculated according to an IPCC tier 2 methodology[52].

Mitigation options for dairy PS

A number of possibilities exist to mitigate GHGE, some concerning the animals, e.g., their genetics for milk yield, but most address management practices. Because of the complex interactions between the various elements of dairy PS and the factors influencing them, any measures intended to reduce the GHGE must be thoroughly examined, as they may exert effects that eventually counteract the intended mitigation.

Feed quality and LUC. One of the most effective strategies to reduce emissions is to increase the energy density of the diet, which usually results in the suggestion to increase the proportion of dietary concentrates. On the one hand, this will lead to a significant decrease of GHGE from enteric fermentation (e.g., -6.5% for PS U_{con} as compared to PS U_{org} if equal milk yields and energy intakes are assumed). On the other hand, increased GHGE from soils and from the use of fertilizers coincide with this mitigating effect and partially counteract it. If an increased use of concentrates is accompanied by LUC (i.e., by converting grassland or pasture into arable land), the mitigating effect of using concentrates turns into an aggravating effect, as shown herein. Another limitation of this practice is the growing probability for digestive and other health disorders associated with increased levels of concentrates in the diet of dairy cows[53,54]. Due to the core role of forages in grassland-based PS (herein, diets were assumed to consist of 76–87% of forage), improving the nutrient density of forage should be prioritized in order to reduce GHGE from enteric fermentation. The greatest effect may be achieved by reducing crude fiber in forage by earlier harvesting (or grazing), but this option is also limited for reasons of grassland ecology[55,56]. According to the calculations conducted within this study, an increase of 0.1 MJ NE_L per kg forage DM will lead to a reduction of total GHGE of about 1.5%.

Lifetime performance. Reducing age at first calving and thereby the rearing phase as well as decreasing the number of lactations in which a constant lifetime performance is yielded would result in a reduced demand for dietary energy and therefore in a reduction of GHGE. On the contrary, a decreased age at first calving and a continuing increase in milk yield per lactation may negatively affect lifetime performance, the number of lactations and the number of offspring per cow[57]. However, an improved lifetime performance (together with a constant or even increasing number of lactations) may be an effective way to decrease GHGE, because emissions from the rearing period will be distributed over a greater amount of milk: GHGE per kg of milk would be reduced by 1.4% on average if lifetime performance was increased by 5000 kg (i.e., from a current value of 23,650 to 28,650 kg for Austria).

Manure management. Furthermore, great potential for reduction of GHGE arises from changes in manure management. The PS in this study were assumed to represent the Austrian situation, where 59.7 and 60.7% of the dairy cows were kept in straw-based systems on conventional and organic farms, respectively[23]. All other cows were kept in systems with slurry production. GHGE from cows' excreta were lower for housing systems with straw litter than for slurry-based systems. A change toward straw-based systems would therefore be desirable because of reduced GHGE and also of a potentially beneficial effect on animal welfare[58]. GHGE per kg milk may be mitigated by 0.9% if a further 10% of dairy barns are changed from slurry- to straw-based systems. However, this is not reflected by the current trend in Austria, where most of the newly built housing systems are slurry based[22]. Moreover, the separation of slurry into solid and liquid phases, as well as an aeration of slurry and composting of farmyard manure would contribute to a mitigation of GHGE, but may lead to increased ammonia emissions[59]. Over all PS covered herein, slurry separation and slurry aeration would reduce total GHGE by 1.8% on average.

When calculating GHGE from manure falling on pasture, only CH_4, but not N_2O, was taken into account, as the latter was already included in the emissions from soil[17]. Overall, pasture-based systems can be considered not only as animal friendly but also as favorable from the point of view of GHGE, as they are emitting less GHG than any other housing systems. On average, cows that spend 10% of their annual time budget on pasture emit 2.4% less GHG as compared to cows that are confined all year round.

Another very substantial reduction potential arises from anaerobic fermentation of manure for biogas production[60], whereby GHGE per kg milk can be decreased by about 5%. Additionally to this direct reduction, the substitution of fossil fuels by biogas could contribute to a further decrease, as 1 Mg of cattle manure can be transformed into 32 kWh of electric energy in a biogas plant[61]. For the PS covered in this study, this would lead to a reduction of total GHGE per kg milk by 7%. Depending on the degree of utilization of the heat that emerges in a biogas plant and the potential substitution of fossil energy for heating, a further potential reduction of GHGE arises.

Utilization of oil seeds for feed and biofuels. The utilization of by-products from the production of biofuels in livestock nutrition is frequently advocated as a contribution to improved sustainability of agricultural production (e.g., UN-Energy[62]). Nevertheless, certain energy crops, such as rapeseed, are frequently reported to cause higher emissions of N_2O than assumed by IPCC and as used in this study and in most previous calculations and reports (e.g., The Royal Society[63])[64]. According to Crutzen et al.[64], the N_2O-emissions from rapeseed are 3–5 times higher than reflected by IPCC[17] default values and current state of life cycle analysis. Assuming five times higher N_2O-emissions for rapeseed, total GHGE from milk would increase by an average of about 2%—or even more in the case of LUC—for the conventional PS.

Reducing energy required for mineral fertilizers and fuels. Besides the production of renewable energy from biogas (as stated in 'Manure management' section), the application of fertilizers and related management measures deserve specific attention concerning their contribution to GHGE. Because emissions from soil increase when the available amount of N in the soil increases[17], the quantity of nitrogen applied must be thoroughly adjusted to the requirements of plants. Furthermore, mineral fertilizers that need large amounts of (fossil) energy during production and transport should be substituted as much as possible by livestock manure. On a long-term basis, housing and manure management systems should therefore be designed in a way that the emissions of nitrogen are kept as low as possible. According to IPCC[17] and calculations herein, pasture can be expected to protect nitrogen very well against emission processes. The use of fossil energy can also be reduced if feeding is generally based on the utilization of pastures and the avoidance of feedstuffs transported over long distances. As an example, the transport of soybeans from Brazil and of extracted soybean meal to be used in PS L_{con} requires 12.3 liters of diesel per cow and year (0.57% of total GHGE).

Conclusions

From the results presented herein and from information provided in the literature, it is concluded that organic milk PS are superior over conventional systems in terms of GHGE both per ha of farmland and per kg of milk. A relevant factor for these differences is LUC as a source for emissions, especially associated with soybean production in South America. For the systems considered in this study, the difference in GHGE per kg of milk between conventional and organic systems depends on the site-specific conditions for agricultural production: the higher the potential milk output per cow, the lower the differences that can be expected.

Regardless of the actual production system, the greatest proportion of GHGE originates from enteric fermentation. Although this inevitable source of emissions can be influenced quantitatively, clear limits exist for the degree of reduction that can be reached. Apart from enteric fermentation, manure management and forage supply also contribute substantially to GHGE. Consequently, dairy PS in which the focus lies on optimum forage quality, a high proportion of pasturage and additional fermentation of the manure in biogas plants will produce relatively low GHGE.

Despite the focus of this paper, the question of sustainable food production should not be restricted to factors that are currently discussed in connection with climate change. However, organic and low-input PS undoubtedly possess a number of strong points with regard to ecological and ethical aspects of sustainability.

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ERRATUM

Greenhouse gas emissions from selected Austrian dairy production systems—model calculations considering the effects of land use change—ERRATUM

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In Figures 1 and 2, a printing error was made concerning the crosshatching for the bar “By-product beef” in the symbol key. The correct crosshatching for “By-product beef” is ▧.

Reference

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2.2 Reduction of greenhouse gas emissions from feed supply chains by utilizing regionally produced protein sources – the case of Austrian dairy production

Research Article

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Reduction of greenhouse gas emissions from feed supply chains by utilizing regionally produced protein sources: the case of Austrian dairy production

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Abstract

BACKGROUND: The aim of this study was to analyse the potential greenhouse gas emissions (GHGE) for regionally alternative produced protein-rich feedstuffs (APRFs) which are utilized for dairy cattle in Austria in comparison to solvent-extracted soybean meal (SBME). In addition to GHGE from agriculture and related upstream supply chains, the effects of land use change were calculated and were included in the results for GHGE. Furthermore, mixtures of APRFs were evaluated which provided energy and utilizable protein equivalent to SBME.

RESULTS: Highest GHGE were estimated for SBME, mainly due to land use change-related emissions. Medium GHGE were found for distillers' dried grains with solubles, for seed cake and solvent-extracted meal from rapeseed and for lucerne cobs. Cake and solvent-extracted meal from sunflower seed as well as faba beans were loaded with lowest GHGE. Substituting SBME by nutritionally equivalent mixtures of APRFs, on average, resulted in a reduction of GHGE of 42% (22–62%).

CONCLUSION: Utilization of locally produced APRFs shows clear advantages in terms of GHGE. Balanced mixtures of APRFs may offer specific benefits, as they allow for a combination of desirable nutritional value and reduced GHGE.

Keywords: dairy cow; protein; anthropogenic greenhouse gas emissions; land use change; carbon footprint; soybean meal

INTRODUCTION

Agriculture, especially animal husbandry, causes considerable greenhouse gas emissions (GHGE). Within livestock husbandry, dairy production systems are the largest source of GHGE.[1] Within these, feeding was found to have a high impact on GHGE.[2] Direct emissions from the feed supply chain account for about 20% of GHGE per dairy cow and year; however, the feeding management exerts a significant effect on emissions from enteric fermentation and from manure. Therefore, total GHGE attributed to feeding are actually higher than stated above. Additionally, GHGE from land use change (LUC) are another source of substantial indirect GHGE connected with feedstuffs.[2]

For several decades, solvent-extracted soybean meal (SBME) has been an important ingredient of livestock diets in Western Europe. Worldwide, soybean is one of the most important plants for human nutrition and livestock feed owing to its high protein and oil contents of 40% and 20%, respectively.[3] Because of agronomic peculiarities such as a relatively low yield and economic disadvantages as compared to other crops, soybean is not cultivated on a large scale in Austria as in other Central and Western European countries.[4]

As a consequence of the increased performance of livestock, the required dietary contents of protein and essential amino acids have also increased substantially over the last decades. Also ruminants, which are able to efficiently convert forage into animal products, have to be fed substantial amounts of concentrates if their productive performance is to be high.[5] Because of its high protein and amino acid contents and the availability of standard technologies for the inactivation of anti-nutritive constituents, SBME possesses a wide range of utilization and is the major protein-rich concentrate in livestock feeding.[6] In the disputed field of ecologically and ethically relevant consequences of the production and import of SBME, two topics are of specific interest besides the scepticism of many European consumers towards the use of genetically modified feedstuffs in livestock nutrition:[4] (1) LUC from grassland, savannah and tropical forest to agricultural land for the production of soybean (and other crops), especially in South America; this LUC is connected with a great loss of carbon in the soils emitted as GHGE (CO_2)[7] and with reduced biodiversity;[8] (2) transport over long distances consume high amounts of energy,

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contribute to GHGE from fossil fuels and render nutrient flows over great distances which counteract attempts to maintain fairly closed nutrient cycles.

Alternative, protein-rich feedstuffs such as grain legumes and by-products from certain oilseeds or grains (cakes, solvent-extracted meals and by-products from distilling) are regionally produced and are used for livestock feeding in Austria, as in other European countries. In the context of ecological sustainability, the question arises whether specific benefits exist for home- or regionally produced, alternative protein-rich feedstuffs (APRFs) in terms of GHGE as compared to SBME. This study therefore assessed the potential benefits of selected APRFs in terms of GHGE which are related to their production and use in dairy cows. Emissions resulting from LUC were specifically emphasized, because previous carbon footprints and life cycle assessments for feedstuffs rarely took this important factor into account.[9,10]

MATERIAL AND METHODS

APRFs and mixtures thereof (APRMs)

Carbon footprints were calculated according to a business-to-business life cycle assessment approach (PAS 2050)[11] for SBME and APRFs as well as for mixtures of these (APRMs) intended to be used in dairy cattle feeding. Additionally, locally cultivated barley was assessed as a reference for energy-rich concentrates.

Eight APRFs which are frequently used in the nutrition of dairy cattle in Austria were estimated according to the GHGE from their supply chain: rapeseed cake (RSC) and solvent-extracted rapeseed meal (RSME), sunflower seed cake (SSC) and solvent-extracted sunflower seed meal (SSME), distillers' dried grains with solubles (DDGS, produced from wheat), lucerne cobs (LC) and faba beans (FB). While LC and FB are used only by relatively few, mainly organic farmers in Austria, RSC and RSME are the most commonly used APRFs by Austrian dairy producers. The nutrient contents of these feed components are characterized in Table 1.

Owing to the high content of crude protein which is utilizable in the duodenum (uCP) and net energy for lactation (NE_L)[12] in SBME, APRFs needed to be mixed and used in greater quantities in order to be nutritionally equivalent to 1 kg of SBME. Herein, mixtures were formulated to represent two different substitution levels for SBME: 50% and 100%, respectively (Table 1); 12 mixtures were formulated which contain the same amounts of NE_L and available protein plus an amount of rumen-undegradable protein (UDP) similar to SBME. These 'equivalent amounts' as given in Table 2 also account for the reducing effect on forage intake of feeding concentrates.[13]

Estimates of feed intake (GfE,[12] Gruber *et al.*[13]) were used for all calculations which included data on feed intake of dairy cows. Within these calculations, forage was assumed to consist of 20% grass, 10% hay, 50% grass silage and 20% maize silage.

Table 1. Selected indicators for the chemical composition of feed components (according to DLG feed tables[19] and Wiedner[20])

	Energy density (MJ NE_L kg^{-1} DM)	Crude protein (g kg^{-1} DM)	Utilizable crude protein (g kg^{-1} DM)	UDP content (g kg^{-1})	Ether extracts (g kg^{-1} DM)	Crude fibre (g kg^{-1} DM)
SBME	8.63	510	288	350	15	67
DDGS	7.41	265	265	400	60	64
RSC	7.99	370	217	300	101	128
RSME	7.20	399	231	250	25	131
SSC	6.53	390	213	300	62	206
SSME	6.25	370	191	250	51	221
LC	5.67	218	184	450	36	222
FB	8.61	298	195	100	16	89

Table 2. Mixtures of protein-rich alternative concentrates (APRMs) which supply NE_L and uCP equivalent to SBME (dry matter basis)

Feed type	Composition of mixtures (g kg^{-1})[a]			Equivalent amount (kg DM)[b]	UDP content (g kg^{-1})
SBME	1000 SBME			1.000	350
APRM 1	467 SBME	394 DDGS	139 RSC	1.105	360
APRM 2	457 SBME	180 DDGS	363 RSME	1.140	320
APRM 3	441 SBME	276 DDGS	283 SSC	1.184	340
APRM 4	444 SBME	178 DDGS	378 SSME	1.196	340
APRM 5	412 SBME	123 DDGS	465 LC	1.316	400
APRM 6	469 SBME	450 DDGS	81 FB	1.100	360
APRM 7		737 DDGS	263 RSC	1.208	370
APRM 8		343 DDGS	657 RSME	1.275	300
APRM 9		320 DDGS	680 SSC	1.367	330
APRM 10		501 DDGS	499 SSME	1.385	330
APRM 11		384 DDGS	86 RSC 530 LC	1.468	420
APRM 12		845 DDGS	155 FB	1.198	360

[a] All mixtures equivalent to 1 kg SBME (8.63 MJ NE_L kg^{-1} DM, 288 g uCP kg^{-1} DM); SBME, soybean meal, solvent-extracted; DDGS, distillers' dried grains with solubles; RSC, rapeseed cake; RSME, rapeseed meal, solvent-extracted; SSC, sunflower seed cake; SSME, sunflower seed meal, solvent-extracted; LC, lucerne cobs; FB, faba beans.
[b] Including the reducing effect on forage intake of feeding concentrates.

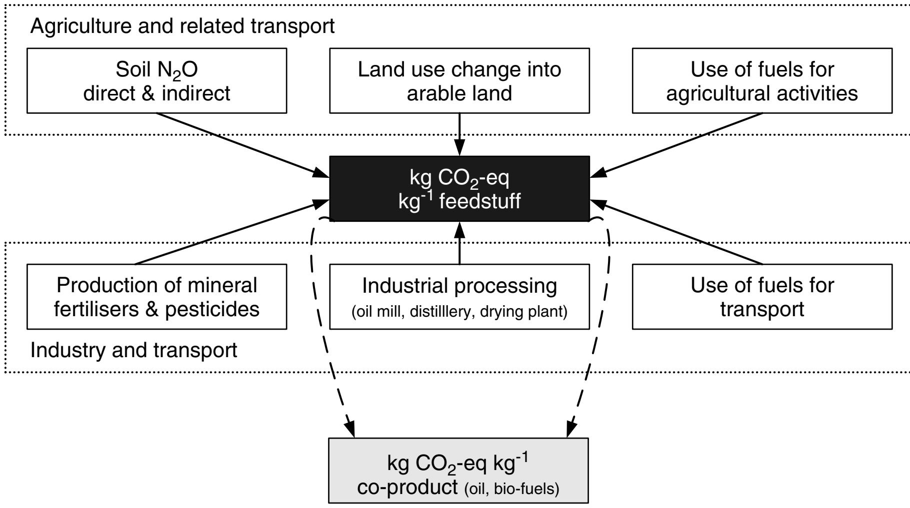

Figure 1. System boundaries for the calculation of GHGE from feedstuffs.

System boundaries, conversion factors and sources of GHGE

System boundaries were defined to include the most important processes leading to GHGE, from the supply of input factors relevant for the production of protein-rich concentrates to the provision of the feed to livestock (see paragraphs below and Fig. 1).

Total emissions were calculated by adding up the emissions of CH_4, N_2O and CO_2 as CO_2 equivalents (CO_2-eq). Conversion factors used to calculate the global warming potential were 25 kg CO_2-eq for 1 kg methane and 298 kg CO_2-eq for 1 kg nitrous oxide (100-year horizon[14]).

In the following paragraphs, the sources of GHGE are described that are considered here.

Agricultural production

Calculation of GHGE generally followed the approach described by Hörtenhuber *et al.*[2] and Lindenthal *et al.*[15] Crop yields per hectare of conventionally managed arable land were derived from Austrian statistical databases;[16–18] energy and nutrient contents of crops and feedstuffs, including UDP content, were taken from feed tables,[19] except for DDGS from wheat.[20] Yields for soybean were derived from Smaling *et al.*[21] and Dalgaard *et al.*[22] for Brazil and Argentina, respectively.

When calculating the GHGE mitigating effect of replacing forage with APRMs (Table 2), values for lucerne–grass mixture were used to represent forage. Table 3 provides information on agricultural production per hectare of farmland for different feedstuffs and for the allocation of emissions (corresponding to caloric values of main products and by-products). GHGE from the use of fuels for agricultural production were calculated according to the ACAERD[23] and Fehrenbach *et al.*[8] Information on transport distances and assumed means of transportation is given in Table 4.

Transport

According to AGES,[4] the vast majority of soy products imported to Austria originate from Brazil (78%), followed by Argentina (20%) and the USA (2%). The soybeans were assumed to be transported by lorries (1000 km of transport) to oil mills near the harbour, where they were processed to oil and SBME and shipped to Europe (10 000 km of waterway).

Fifty-five per cent of rapeseed processed into oil and RSC in Austrian oil mills were cultivated in the region, whereas the other

Table 3. Data on agricultural production per hectare of arable land and on caloric allocation of emissions

	Barley	SBME[a]	DDGS[b]	RSC	RSME	SSC	SSME	LC	FB
Yield – feedstuff (kg DM ha^{-1})	4800	2040	1891	2205	1953	1668	1575	8000	2670
Yield – co-product (oil/bio-ethanol; kg ha^{-1})	0	510	1,891	945	1,197	852	945	0	0
Caloric value – feedstuff (%)	100.0	67.2	45.0	56.5	45.4	43.1	37.6	100.0	100.0
Caloric value – co-product (oil/bio-ethanol; %)	0	32.8	55.0	43.5	54.6	56.9	62.4	0	0
Fuels – agriculture (l ha^{-1})	71.5	60.0	71.5	73.0	73.0	64.0	64.0	76.0	67.0
Fertilizer input – nitrogen (kg N ha^{-1})	88	0	106.9	133.4	133.4	33.3	33.3	0	0
Fertilizer input – phosphorus (kg P_2O_5 ha^{-1})	38.0	16.0	40.7	56.7	56.7	45.4	45.4	61.0	26.5
Fertilizer input – potassium (kg K_2O ha^{-1})	34.0	0	34.6	44.1	44.1	42.3	42.3	193.0	44.5
Fertilizer input – limestone (kg CaO ha^{-1})	300	300	300	300	300	300	300	100	300
Pesticide input (kg pesticides ha^{-1})	3.5	3.0	3.8	2.5	2.5	2.0	2.0	0.0	5.1
Fuels – transports of inputs (kg fuels ha^{-1})	5.13	8.82	5.38	5.94	5.94	4.68	4.68	3.92	4.16
N in soils (kg N ha^{-1})	108.0	50.0	126.9	174.4	174.4	63.3	63.3	80.0	80.0

[a] Imported from South America.
[b] From wheat.

Table 4. Data on demand for fuels for agriculture and transports per kg (DM) feedstuff

	Barley	SBME	DDGS	RSC	RSME	SSC	SSME	LC	FB
Fuels – agriculture (L fuels kg^{-1} DM feedstuff)	0.016	0.020	0.018	0.019	0.017	0.017	0.016	0.010	0.025
Transport distance – inputs (km)	200	500	200	200	200	200	200	200	200
Transport vehicles – inputs	Lorry	Lorry	Lorry	Lorry	Lorry	Lorry	Lorry	Lorry	Lorry
Transport amount – inputs (kg)	464	319	486	537	537	423	423	354	376,1
Fuels – transport inputs (L fuels kg^{-1} DM feedstuff)	0.001	0.002	0.001	0.002	0.001	0.001	0.001	0.000	0.002
Transport distance – crops/feedstuff step I (km)[a]	50	1000	375	275	325	275	325	50	50
Transport vehicles – crops/feedstuff step I	Tractor	Lorry	Lorry	Lorry	Lorry	Lorry	Lorry	Tractor	Tractor
Fuels – transport crops/feedstuff step I (L fuels kg^{-1} DM feedstuff)	0.005	0.026	0.021	0.014	0.015	0.011	0.012	0.025	0.005
Transport distance – crops/feedstuff step II (km)[b]	50/10	10,000/700/700/10	150/10	50/10	150/10	50/10	150/10	150/10	50/10
Transport vehicles – crops/feedstuff step II	Lorry/tractor	Ship/train/lorry/tractor	Lorry/tractor	Lorry/tractor	Lorry/tractor	Lorry/tractor	Lorry/tractor	Lorry/tractor	Lorry/tractor
Fuels – transport crops/feed step II (L fuels kg^{-1} DM feedstuff)	0.004	0.087	0.010	0.004	0.010	0.004	0.010	0.010	0.004
Total fuels – agriculture and transports (L fuels kg^{-1} DM feedstuff)	0.025	0.136	0.049	0.038	0.037	0.033	0.039	0.045	0.036

[a] From the farm to a reloading point, to a processing industry or to a point of sale.
[b] From the reloading point via/from a processing industry or from a point of sale to a farm.

45% were imported, mainly from Eastern and Central Europe (28% Hungary, 15% Slovakia, 2% Croatia).[24] The latter results in an average 275 km transport by lorry. Generally, transport distances were split into two steps: (1) from the farm to the industry (for processing, e.g. oil mill), to a reloading point or to a point of sale; (2) from the industry or a reloading point via a store house (point of sale) to a farm by lorry, ship or train and a tractor, as shown in Table 4. However, the transport distances between oil mill or store house and farm were relatively short (50 km by lorry plus 10 km by tractor). Generally, transport distances accounted both for the outward freight and for the return. For the return journey the same distance as for the outward freight was accounted for by tractors, but only half the distance for transport by lorry due to the usual practice of transporting outward freight.

RSME as well as DDGS and LC are produced only in one location in Austria, which leads to higher transport distances between processing plant and farm (150 km by lorry and 10 km by tractor).

Table 4 summarizes information on the demand for fuel of agriculture and transport per kilogram (dry matter, DM) of feedstuff. GHGE from the use of fuel for transports were estimated according to Wilting *et al*.[25] and Fehrenbach *et al*.[8]

Mineral fertilizer, pesticides and emissions from soil

Information on mineral fertilizers and pesticides for South American soybean production was taken from Dalgaard *et al*.[22] Amounts of pesticides and fertilizers (N/P/K/Ca) applied per hectare of arable land were derived from Austrian statistical databases.[18] GHGE from the production of mineral fertilizers and pesticides were estimated based on information given by Patyk and Reinhardt[26] and Biskupek *et al*.,[27] respectively. In addition to direct N_2O emissions from soils – which corresponded to the amount of N applied as fertilizers – N from atmospheric deposition and from crop residues was considered for indirect N_2O emissions according to IPCC (International Panel on Climate Change) guidelines.[7] Additional indirect emissions of N_2O from leaching were calculated following IPCC guidelines,[7] using a default value for leached N of 30% of total soil N.

Owing to missing values, data derived for Austrian conditions on N input via atmospheric deposition[28] and on N input via crop residues of soybean[29] were assumed for calculating soil N pools for South American soybean.

Industrial processing of feedstuffs and allocation of GHGE to products

One kilogram of soybean was assumed to be transformed into 0.18 kg oil and 0.80 kg SBME, with 0.02 kg of loss.[22] One kilogram of rapeseed is transformed into 0.30 kg oil and 0.70 RSC,[30] and 1 kg sunflower seeds into 0.34 kg and 0.38 kg oil if pressed and solvent-extracted, respectively, the residual amounts being SSC and SSME, respectively.[31] According to Vetter *et al*.,[32] bio-ethanol and DDGS are produced at a ratio of 50 : 50 in a distillery, with an efficiency of about 0.34 kg each per kilogram of wheat. Based on these numbers and on the energy content of the products,[8,19] GHGE were allocated to the individual (by-)product.

GHGE from industrial processing of feedstuffs were adopted from Fehrenbach *et al*.[8] for the bio-ethanol distillery and from Lehuger *et al*.[33] for oil mills. GHGE from processing of LC were calculated according to data from Nielsen[34] and Austrian emission factors for use of energy according to Ecoinvent.[35]

Land use change (LUC)

According to IPCC guidelines,[7] GHGE resulting from LUC were considered for areas where land was converted during the last 20 years (since 1990). LUC is a major source of GHGE if conventionally produced SBME is imported for feeding purposes from South America.[2] According to FAO 2008 statistics[36] for Brazil, the increase in total arable land was found to be mainly connected with a loss in tropical forest area and to a lesser extent with a loss of savannah during the last 20 years. As production of soybean is the major driver for LUC in Brazil,[21] the conversion factors for total arable land were applied for soybean production. Overall, 52% of land for cultivation of soybeans has not been subject to LUC for the last 20 years, 39% have been changed from tropical forest and 9% from savannah. However, a remarkable part of SBME imported to Austria has to be certified as being GMO-free or has to be replaced by APRFs and APRMs, as about 57% of total milk produced in Austria in 2007 was found to be sold as 'GMO free' in order to meet consumers' demands. The respective certification criteria also include regulations about LUC from tropical forest areas to agricultural land ('Basel criteria').[37] Therefore, fewer GHGE from LUC were attributed to SBME typically used in diets for dairy cattle in Austria ('dairy SBME'). Based on the FAO 2008 statistics[36] and certification criteria,[37] it was assumed that for certified SBME about 88% of the respective Brazilian soybean fields had not been subject to LUC and that 10% of the areas were changed from tropical forests to agricultural land and 2% from savannah to agricultural land during the last 20 years. For Argentina, it was assumed that 95% had already been fields before 1990 and that 5% had been changed from savannah to agricultural land according to the data from FAO.[36] GHGE from this LUC in South America were calculated based on data reported by RFA (Office of the Renewable Fuels Agency, Department for Transport).[38] Consequently, taking into account the proportion of imports and country of origin, LUC-related GHGE of 2.665 kg CO_2-eq were calculated for an average kilogram of dairy SBME (mixture of certified and non-certified SBME) in addition to GHGE from cultivation, transport and processing. As supported by data from the European Environment Agency (EEA),[39] LUC from conversion of grassland into arable land could be neglected for Austria, but had to be considered for imports of rapeseed from Eastern and Central Europe. GHGE from LUC for imported rapeseed and wheat were estimated based on Fehrenbach *et al*.,[8] combined with data from EEA[39] and data on imports (as given under 'Transports' above). FB, LC and barley were assumed not to be related to LUC as they are integrated in the crop rotation of regional farms and an increase in their production – which could potentially lead to LUC – is highly unlikely.

RESULTS

GHGE for supplying APRFs

The supply of protein-rich concentrates is linked to the emission of different amounts of greenhouse gases as described in Table 5: 1 kg of dairy SBME (DM) was found to be loaded with the highest GHGE of 3.278 kg CO_2-eq if LUC was taken into consideration; if soybeans were produced without LUC, GHGE would be reduced to 0.613 kg CO_2-eq. Faba beans as well as cake and solvent-extracted meal from sunflower seeds showed the lowest GHGE per kilogram DM, ranging from 0.372 to 0.679 kg CO_2-eq if LUC occurred and from 0.300 to 0.445 without LUC. Relatively high GHGE were found for LC and for DDGS, with 0.915 and 1.450 kg CO_2-eq kg^{-1} DM, respectively, if LUC occurred. Without taking LUC into account, DDGS resulted in the highest GHGE of all protein-rich concentrates (1.191 kg CO_2-eq kg^{-1} DM), mainly due to GHGE related to industrial processes. Cake and solvent-extracted meal

Table 5. Resulting emissions (GHGE) from fuels, industrial processes, the production of mineral fertilizers and pesticides, from direct and indirect N_2O and land use change per kilogram of feedstuff DM

	Barley	Dairy SBME[a]	DDGS	RSC	RSME	SSC	SSME	LC	FB
GHGE from fuels (kg CO_2-eq kg^{-1} feedstuff DM)	0.084	0.453	0.164	0.126	0.123	0.109	0.128	0.151	0.120
GHGE from industrial processes (oil mill, distillery, drying plant; kg CO_2-eq kg^{-1} feedstuff- M)	0	0.050	0.748	0.035	0.034	0.035	0.034	0.681	0
GHGE from production of mineral fertilizers and pesticides (kg CO_2-eq kg^{-1} feedstuff-DM)	0.163	0.018	0.111	0.206	0.165	0.065	0.056	0.027	0.046
GHGE from agriculture – N_2O from soils (direct and indirect; kg CO_2-eq kg^{-1} feedstuff DM)	0.125	0.092	0.168	0.249	0.226	0.091	0.084	0.056	0.279
GHGE from LUC (kg CO_2-eq kg^{-1} feedstuff DM)	0	2.665	0.259	0.397	0.319	0.379	0.330	0	0
Total GHGE without LUC (kg CO_2-eq kg^{-1} feedstuff DM)	0.372	0.613	1.191	0.616	0.548	0.300	0.302	0.915	0.445
Total GHGE with LUC (kg CO_2-eq kg^{-1} feedstuff-DM)	0.372	3.278	1.450	1.013	0.867	0.679	0.632	0.915	0.445

[a] Representing the average Austrian dairy production system, where about 50% of SBME is assumed to be certified (GMO-free and less LUC).

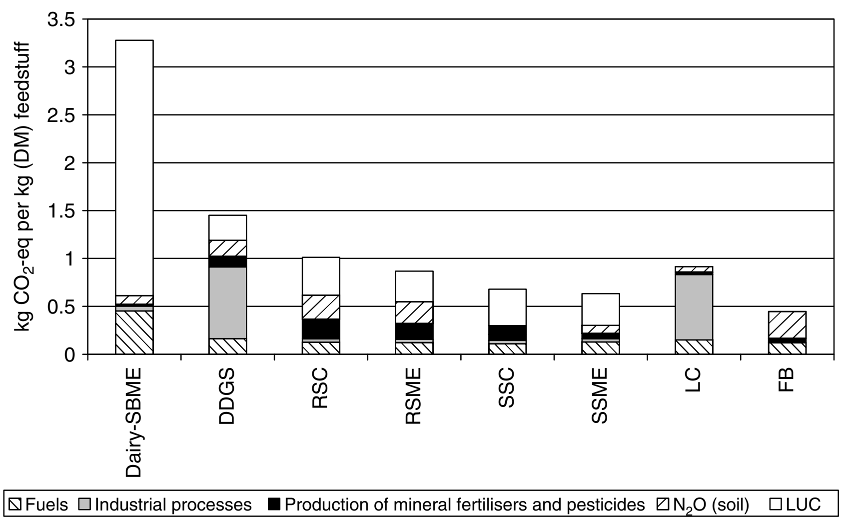

Figure 2. GHGE linked to the supply of different protein-rich feedstuffs (kg CO_2-eq per kilogram of feedstuff DM).

from rapeseed ranged between products from sunflower seeds and DDGS. GHGE for RSC were found to be 1.013 kg CO_2-eq and 0.616 with and without LUC, respectively. The provision of RSME resulted in 0.867 kg CO_2-eq kg^{-1} DM if LUC occurred and 0.548 kg CO_2-eq kg^{-1} DM without LUC. Figure 2 illustrates the results for GHGE of protein-rich feedstuffs and the contributions from different sources.

Land use change (LUC)

LUC plays a central role for the GHGE which are related to the use of the feedstuffs analysed in this study: LUC contributed 18% to total GHGE for DDGS, while it plays a much greater role (up to 81%) in the case of SBME. For most feedstuffs LUC was responsible for the highest proportion of GHGE, except for barley, LC and FB, which were assumed not to be related to LUC. Specifically, high GHGE from LUC occur for SBME which was produced in South America (Table 5).

Mineral fertilizer and emissions from soil

RSC and RSME showed the highest GHGE from mineral fertilizer and pesticide production, as well as soil N_2O emissions. The relative contribution of mineral fertilizer and pesticide production to overall GHGE varied from 0.5% for South American SBME to 19% and 20% for RSME and RSC, respectively (Fig. 2 and Table 5). On average, direct and indirect N_2O emissions from soil accounted for 25%, ranging from 3% for SBME to 63% for FB. The relative contribution of N_2O to GHGE was found to be about 25% for RSC and RSME.

Energy used as fuels and for industrial processing

For all APRFs, fuels for transport and agricultural production on average accounted for only 16% of GHGE. However, in absolute numbers, GHGE from fuel consumption are high for SBME due to the long transport distance as compared to regionally produced feedstuffs. For all feedstuffs considered here, GHGE from transport on average accounted for 57% of total GHGE from fuels, with a range from 24% (FB) to 84% (SBME); the rest was related to agricultural activities (43% on average).

Except for LC and DDGS, where the shares of GHGE from industrial processing were found to be high (74% and 52%, respectively), industrial processes accounted for only 2–5% (e.g. from pressing and extraction of oil seeds).

Mitigation of overall GHGE by using APRMs as a substitute for SBME

A replacement of SBME by mixtures of APRFs decreases GHGE significantly (Fig. 3). A replacement rate of about 50% (APRM 1–6) decreases GHGE by about 26% (22–29%) in comparison to dairy SBME. A complete substitution of SBME (APRM 7 and 12) decreases GHGE on average by about 55% (49–61%) in comparison to dairy SBME as currently used in Austria. Figure 3 shows GHGE of the 12 APRMs as compared to 1 kg SBME. These calculations account for both the reduction of forage intake through concentrate feeding and the mitigated GHGE from reduced forage intake. As shown in Table 6 for exemplary diets, GHGE which are attributed to concentrate supplementation increase with increasing daily milk yield.

DISCUSSION

Relevant sources for GHGE

Specifically high GHGE from LUC occurred for SBME which had been produced in South America (Table 5). Rainforest clearance and the ploughing of savannahs – the latter with a lesser effect – result in high CO_2 emissions from the burning of huge amounts of organic material and from the reduction of organic carbon stocks by mineralization in agricultural soils.[7,8] GHGE are much lower for APRMs as compared to SBME (Table 5). Nevertheless, the greatest proportion of GHGE associated with the production of most APRMs was also found to be attributed to LUC.

If no LUC occurred, GHGE would be highest for DDGS due to the industrial processes involved in their production; these high GHGE from industrial processes in DDGS reflect their unfavourable energy balance. GHGE would be much more favourable for SBME and would be similar to those for other feedstuffs, if it was produced without converting grassland or forests into arable land (LUC). As supported by Lehuger *et al.*,[33] GHGE from SBME would then be similar to or even below those from RSC. If LUC and transport could be reduced by growing soybean locally or by importing them from nearby countries of Southern and Eastern Europe, SBME would come off even better, mainly due to its biological N fixation (i.e. less mineral N fertilizer required[33]).

However, LUC was not always considered in previous estimations of carbon footprints,[9,10] but is assumed to contribute up to nearly one-fifth to anthropogenic GHGE.[14] Especially where forest clearing occurs in the Tropics, LUC will be the major source of GHGE and should therefore be introduced into estimations of GHGE from food supply chains. This is of particular relevance for the scenarios covered here, as the vast majority of SBME imported into Austria is assumed to be connected to a certain degree of LUC in the countries of origin.[8]

In contrast to Austrian dairy SBME, for the Austrian average of non-certified SBME only 61% of production areas were assumed

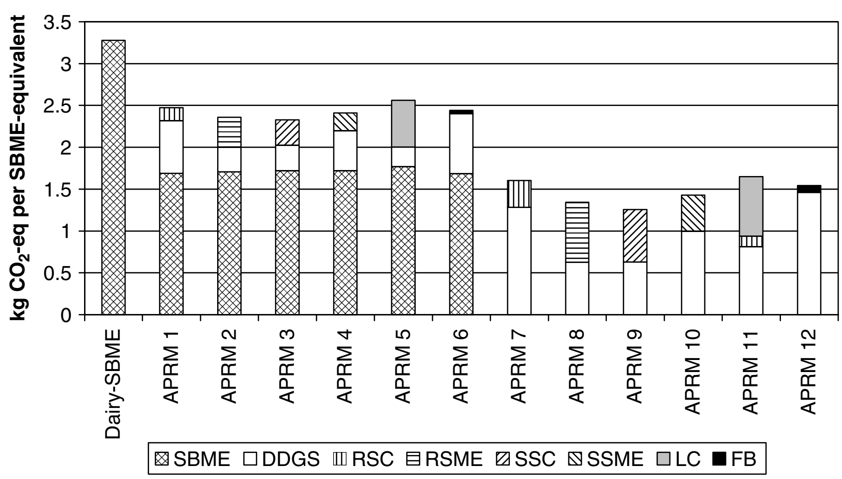

Figure 3. GHGE of SBME and 12 mixtures of APRFs supplying NE_L and utilizable protein equivalent to SBME (kg CO_2-eq per SBME equivalent).

Table 6. Exemplary calculations for GHGE from concentrate supply using APRFs and APRMs in a typical Austrian forage-based diet

	Daily milk yield (kg)[a]				
	15	20	25	30	35
Forage intake (kg DM d^{-1})	12.5	12.9	13.4	13.9	14.4
Barley (kg DM d^{-1})	0.00	2.96	4.15	5.42	6.76
Type of protein concentrate	FB	FB + APRM 9	APRM 8	APRM 3	APRM 1
Amount of protein concentrate (kg DM d^{-1})	1.85	0.37 + 0.51	1.33	1.61	1.87
GHGE from concentrates per day (kg CO_2-eq)	0.82	1.73	2.96	5.30	6.70
GHGE from concentrates per kg milk (kg CO_2-eq)	0.055	0.087	0.118	0.177	0.192

[a] Containing 4.1% fat and 3.4% protein.

not to be related to LUC emissions, 8% were converted from savannah and 31% were converted from forest. While non-certified SBME would in total result in 4.763 kg CO_2-eq kg^{-1} SBME (DM), including 4.150 kg CO_2-eq from LUC, 1 kg (DM) of SBME certified according to 'Basel Criteria'[37] shows a total GHGE of 1.828, including 1.245 kg CO_2-eq from LUC. Because of the relevance of certified SBME in Austrian dairy production, 1 kg dairy SBME on average results in 3.278 kg CO_2-eq with 2.665 kg CO_2-eq from LUC. Consequently, the advantage of a substitution of SBME by APRMs will be even greater in the case of non-certified SBME.

In general, results for GHGE per kilogram of product presented here should not be directly compared to results for the global warming potential from other studies (e.g. Williams *et al*.,[40] Dalgaard *et al*.[22]), as LUC was introduced as an additional source for GHGE herein, but also due to different allocation methods used, different transport distances or agricultural practices presumed. Despite these differences, the estimated GHGE from feedstuffs correspond in magnitude to values and trends given by other authors: e.g. Dalgaard *et al*.[22] estimated GHGE from LUC to be about 5 kg kg^{-1} soybean, which is quite similar to the 4.15 kg for a non-certified SBME imported to Austria. GHGE from RSME without accounting for LUC were found to be 0.548 kg CO_2-eq kg^{-1} feedstuff, which are very close to the 0.550 kg reported by Williams *et al*.[40] The difference between these values to the total GHGE of 0.867 kg (Table 5 and Fig. 2) emphasize the importance of properly accounting for LUC.

The production of mineral N fertilizers consumes large amounts of energy and emits N_2O during production, but also the N applied to soils leads to N_2O emissions. Therefore total GHGE from crop production are closely connected to the amount of fertilizers applied. Rapeseed usually needs large amounts of mineral N (as well as P and K) fertilizers and pesticides and therefore showed high GHGE. The relative proportion of GHGE from soil N_2O was also high for FB, although no N fertilizers were used for their production. The reasons for this are relatively low GHGE from other sources (e.g. short transport, no LUC), low yields, a relatively high amount of N left as crop residues and the lack of co-products which GHGE could be partially attributed to. However, residues of FB leave N in the soil and therefore allow for a reduction of mineral fertilizers to be applied in the following year(s), thereby exerting a mitigating effect of about −0.168 kg CO_2-eq kg^{-1} (DM) of FB. If this effect was accounted for, GHGE for FB would be 0.277 kg CO_2-eq kg^{-1} (DM).

Generally, where nutrient cycles are closed more completely or where less nutrients circulate in the system, gaseous N emissions (N_2O) are potentially lower.[41] Similarly, the effect of biological N fixation can help to lower GHGE from a crop rotation. If (protein-rich) concentrates are cultivated on-farm, the GHGE balance of feeds will be improved not only because of reduced transports but also due to the recycling of cattle excreta as fertilizer and hence more intact nutrient cycles. Thereby, substantial amounts of GHGE (10% on average, with peak values of 20% for RSC) can be mitigated if manure replaces mineral fertilizers.

If GHGE are completely loaded on the major product from a production process, emissions for by-products would be theoretically set to zero. Such an approach would severely bias the assessment of the global warming potential of input factors which are by-products from upstream production processes. Therefore, in this study GHGE were allocated proportionally to products and by-products, depending on their energy contents. As compared to an economic value-based allocation, the approach used here may eventually result in a specific advantage for feedstuffs which are by-products from manufacturing of energy-rich products; e.g. for DDGS the co-product ethanol is loaded with a greater share due to its higher caloric value; subsequently DDGS come off relatively well if caloric allocation is used.

Practical relevance

The nutrient density of protein feedstuffs must increase with increasing milk yields; GHGE from concentrates (barley plus APRFs/APRMs; Table 6) also rise as the performance of dairy cows increases. On the other hand, GHGE from other sources mostly decrease with a higher milk yield per cow (e.g. GHGE related to energy required for maintenance or to the rearing phase).[2] Besides increasing milk yields, a decreasing forage quality would also have to be compensated for by an increased dietary proportion of concentrates; this would again lead to increasing GHGE per kilogram of milk. Consequently, reducing the dietary proportion of concentrates by increasing the nutritional value of forages on the one hand is an important mitigation option.[2] On the other hand, this allows an even greater use of APRFs and APRMs with a specifically low GHGE load, such as FB or APRMs which contain SSC or SSME.

For some of the APRFs, upper limits have to be considered for their inclusion in the diets of dairy cows. The need for such a limitation may be due to several factors, among others constituents which reduce the acceptance of the feed by livestock (e.g. tannins), high fibre content (e.g. for by-products from sunflower seeds[6,42] or LC) or a low content of important nutrients (e.g. UDP).[6] If such feedstuffs were used in mixtures (APRMs), this issue would be of less relevance due to 'dilutive' or 'compensatory effects'. Conversely, some APRMs such as APRM 5 and APRM 11, both containing high amounts of LC, may possess specific advantages because of their high UDP content of 40% and 42%, respectively. APRMs containing relatively high proportions of DDGS, such as APRM 1, APRM 6 and APRM 7, may be similarly advantageous.

The different APRMs formulated here (Table 2) can be specifically recommended for the supplementation of diets based on forages and energy-rich concentrates for different levels of milk yield: APRMs 1–6 and 8 may substitute SBME for daily milk yields up to at least 40 kg; the other APRMs are suitable for milk yields of about 30 kg per day. The use of APRMs 4 and 10, which contain SSME, may be limited in cases of particularly high energy requirements.[6] Similar limitations apply to APRMs 5 and 11, containing LC. The reduced substitution rate of 50% (APRMs 1–6) most likely applies to diets for high yielding dairy cows, which require large amounts of protein. However, all 12 APRMs allow for a relatively high level of performance. APRMs 7–12, especially, demonstrate the great potential of mitigation effects in connection with a change in feeding regime.

When discussing scenarios for the substitution of SBME in livestock production, it should be noted that replacing SBME by APRFs or APRMs is much easier for ruminants than for monogastric animals. For Austria, it was estimated that in 2006 about 120 000 t out of 600 000 t of imported SBME were fed to cattle, about half of this amount being used for dairy cattle and their offspring.[6] APRMs 7–12 are considered suitable substitutes for SBME for the majority of Austrian dairy farms, where cows typically produce at a medium level of performance (20–30 kg per day) for long periods throughout the lactation.[43] Farms which operate high-output production systems may choose to feed APRMs 1–6, which still contain some SBME, in order to maintain a higher nutrient and energy density as compared to APRMs 7–12. Similar considerations may be generally relevant for high-yielding dairy cows in early lactation.

Based on data for the availability of protein sources in Austria[4,6,41] and on the scenarios described here, it can be expected that DDGS may represent up to 55% of all components present in APRMs for dairy cattle, followed by SBME (14%), RSC (13%) and RSME (11%). FB, LC, SSC and SSME are likely be utilized at much smaller rates of about 1–2%.

Consequently, the amount of SBME currently required for feeding Austrian dairy cattle could be reduced by about 83% (i.e. from 60 Gg (gigagrams) to about 8.6 Gg per year). However, about 33.2 Gg DDGS, 7.9 Gg RSC, 6.6 Gg RSME, 0.9 Gg SSC, 1.1 Gg SSME, 0.5 Gg LC and 1.1 Gg FB will have to be additionally produced and processed if this potential substitution rate is to be achieved. The mitigation potential of this substitution would be about 53% of GHGE (−104.4 Gg CO_2-eq).

In Austria, a partial substitution of SBME by APRFs in commercially produced compound feed already started some years ago[44] because of an increasing demand for products from GMO-free dairy production and the rising costs of SBME on global feed markets. Other than commercial feed mills, farmers who produce concentrate mixtures on-farm face greater difficulties in implementing such a substitution strategy, partly due to the greater variability in the nutritive value of APRFs as compared to SBME.[44]

However, it has to be kept in mind that the implementation of such a substitution scenario may exert a driving force for LUC in Europe, as the potential for an increased production of APRFs is limited on the regionally available agricultural land area. In this case, the effects of LUC in Europe related to increased production of APRFs need to be assessed relative to reduced LUC in South America. Future studies should therefore include a wide range of effects which are related to feed supply options, among others ecological consequences and ethical implications such as a potential competition with food supply for humans.[10,45]

CONCLUSION

Because of the LUC-related high GHGE of SBME originating from South America, a partial or complete substitution of SBME by regionally produced, protein-rich concentrates offers an important option for mitigating GHGE from dairy production systems. Formulating mixtures from regionally produced feedstuffs maintains a high nutritive value, while at the same time significantly reducing GHGE from the supply chains of protein-rich concentrates.

LUC may also be relevant – although to a much lesser degree than for SBME – in the production of alternative concentrates. The relevance of single sources of GHGE is quite different for different APRFs: although quantitatively varying, the most important sources are LUC (for RSC, RSME, SSC and SSME), industrial processes (for DDGS and LC) and N_2O emissions from the soil (for FB). This calls for a thorough analysis of GHGE in the assessment of environmental effects of different feed supply options and for the identification of the most important sources of GHGE in order to define strategies for their reduction.

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2.3 Land Use and Land Use Change in Supply Chains for Food and Feedstuffs – Background, Methods and Impact for Life Cycle Assessments and Carbon Footprints

Land Use and Land Use Change in Supply Chains for Food and Feedstuffs – Background, Methods and Impact for Life Cycle Assessments and Carbon Footprints

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Abstract

Investigating the supply chain of the product allows to understand its environmental impacts. In all agricultural activities, land use change (LUC) is one of the major contributors to global CO_2-emissions. Due to methodological difficulties, however, this factor is rarely considered for estimations of greenhouse gas (GHG) emissions from supply chains for food and feedstuffs. We provide here a consistent methodology to allow coverage of LUC-related CO_2-emissions as well as their removal from atmosphere, and we propose to strictly focus on effective (physical) fluxes of GHGs which occur in connection with the supply chain of a product. Simulations of atmospheric CO_2 concentrations allow to determine the impact of LUC since pre-industrial times. The results show that almost a quarter (23%) of the increase in CO_2 concentration which occurred during the last 250 years comes from LUC and – to a much lesser extent – from land use (LU), with a rather drastic impact of LUC occurring during the last few decades. Thus, CO_2 emitted from both soil and vegetation due to LU and especially occurring after LUC shall be considered within accounting periods of ten or 20 years. The length of these accounting periods is consistent with the timescale of soil carbon losses from isolated areas which remain in the atmosphere. For the case of Brazilian soybeans, LUC and LU considered over that period lead to emissions of 5.2 to 6.7 kg CO_2 and 0.1 kg CO_2 per kg product (dry matter), respectively.

Key Words

Land use change, atmospheric CO_2, emission, feedstuff, life cycle assessment, LCA, carbon footprint

1 Introduction

Agriculture, and especially animal husbandry, cause considerable greenhouse gas (GHG) emissions. In addition to the emissions of the livestock themselves, also the production of feed has to be considered and was found to have a high impact on GHGs from direct emissions (soil or fuels) and indirect emissions such as land use change (Hörtenhuber et al., 2011). In this paper, we specifically address the consequences on GHGs originating from the production of feedstuffs. Land use change (LUC, also termed 'land conversion' or 'land transformation' in life cycle assessment) from grasslands, savannahs or forests to agricultural land, especially when it occurs in the tropical regions of South-America, Asia and Africa, is generally assumed to be one of the major contributors to global CO_2-emissions (e.g. Houghton, 2008; Denman et al., 2007). Emissions from LUC are expected to contribute about 20 % of total global CO_2-emissions for the 1980ies and 1990ies (Denman et al., 2007). On the one hand, LUC is connected with a great loss of carbon from aboveground biomass. On the other hand, soil organic carbon (C_{org}) is mineralized and emitted as a consequence of LUC and land use (LU, also termed 'land occupation'), contributing to GHG emissions mainly in the form of CO_2. Furthermore, LUC causes other negative effects on sustainability, as it usually results in a significantly reduced biodiversity, especially in tropical regions (e.g. ten Brink, 2009) and a loss of water in the global water cycle (e.g. Avissar and Werth, 2005).

Despite its great impact on global GHG emissions and thus on global warming, (direct) LUC is hardly incorporated into the global warming potential (GWP, as an indicator for life cycle impact on GHG emissions) and is rarely taken into account in current carbon footprints (CFs) in assessments for food and feedstuffs (Garnett, 2009; de Vries and de Boer, 2009) for various reasons; these include: (a) conceptual and (b) methodological limitations (e.g. Dalgaard et al., 2008). Conceptual limitations resulted in LUC not being explicitly covered by Dalgaard et al. (2008), partially due to difficulties in allocating them to the appropriate functional unit and the question of whether or not to include above-ground-(biomass-) and soil-emissions. Furthermore, calculations of GHG emissions must be seen as problematic (following Daalgard et al., 2008) if LULUC (land use and land use change) emissions are considered only for some inputs, such as imported feedstuffs from Brazil or Argentina, but not consistently for all factors leading to carbon-sequestration or -emission, e.g. for inputs produced on European farmland. In this context, Dalgaard et al. (2008) cites a number of publications, which do not (fully) take C_{org} fluxes into account (e.g. Basset-Mens and van der Werf, 2005). Concerning methodological limitations, Dalgaard et al. (2008) refer to the unknown nature of land (use) before conversion occurred to another land use category and they also point out that the quantitative changes in above-ground and below-ground carbon are mostly unknown. Additionally, the amortisation period is debatable and consequently it is not clear whether emissions should be completely ascribed to the crops cultivated during the first year or distributed over a disputable period of years of cultivation (Dalgaard et al., 2008).

As a result of the conceptual and methodological limitations mentioned above, in many previous studies system boundaries are defined rather narrowly and GHG emissions related to carbon-sequestration or -emission from LU and LUC are excluded. Subsequently, not all fluxes of carbon and other elements are considered in the studies cited above. Therefore the question arises if a CF or a GWP exhibit the full impact unless they cover all relevant flows.

Assessing the life cycle of agro-biofuels, a number of studies already included GHG emissions from (LU)LUC due to a detailed description of carbon-cycles, before they were included into estimations of GHG emissions from food supply chains (e.g. Fargione et al., 2008; Searchinger et al., 2009). The issue is critical for biofuels as the key argument for their production is that they 'save CO_2'; thus the argument needs to be scrutinized. Since this applies to agricultural production in general, carbon emission or sequestration from LU and LUC might similarly be considered in feedstuff and food supply chains, especially as parts of their ingredients are derived as co-products from production of agro-fuels.

In recent years, few guidelines and studies concerning estimations of GHG emissions from food supply chains tried to include LU- and LUC-emissions, but sometimes relied on different methods for their calculation: Guidelines published by the British Standards Institution in a 'Publicly Available Specification' (PAS2050, 2008) and standards presented by the World Business Council for Sustainable Development and the World Resources Institute (WRI/WBCSD, 2009), or the quite contrasting method described by Kool et al. (2009) may serve as examples.

A literature review shows that only few authors attempted to assess LULUC-related contributions to GHG emissions, among them publications by Müller-Wenk and Brandão (2010) on LUC in LCAs and by Searchinger et al. (2008) as well as Fargione (2008) on LUC and agro-fuels. In the vast majority of publications no proper consideration of LUC was found when estimating GHG emissions.

This identifies a need to improve methods suggested in previous literature, e.g. PAS2050 (2008) as well as the standards presented by WRI/WBCSD (2009). Therefore this paper focuses on suggesting a method based on a sound consideration of LUC-related CO_2-emissions as well as CO_2-removal from the atmosphere.

The aims of this paper are: (1) To contribute to the ongoing debate about a well-founded consideration of LU and LUC in terms of GHG emissions with special attention on (a) system boundaries, (b) sound accounting periods for LUC and (c) the potential inclusion of GHG emissions from biomass or C_{org}, which are often propagated to be 'CO_2-neutral'. (2) To describe a method which could be utilised for an estimation of emissions from LU and LUC, and (3) to derive effects of different approximations on GHG emissions from LULUC for supply chains for food and feedstuffs.

2 Material and Methods

2.1 Simulating atmospheric CO_2 concentrations

In order to adequately simulate the consequence of food and feed production to the atmosphere this paper uses established methods to link emissions of CO_2 to resulting atmospheric concentrations.

The atmospheric lifetime of a compound is defined as the time to reach 1/*e* (about 37 %) of the initial amount. Atmospheric reactions or sinks determine the removal of the respective compounds. Methane (CH_4) is known to have a lifetime of 12 years, dinitrous oxide (N_2O) of about 120 years (Forster et al., 2007). The situation is more complex for CO_2. CO_2 is removed from the atmosphere to the oceans (different oceanic pools provide different removal characteristics) and to terrestrial

sinks (e.g. growth of vegetation and accumulation in C_{org}). Therefore, no single average lifetime can be derived for CO_2 (see e.g. Lashof and Ahuja, 1990). 21.7 % of CO_2 is even assumed to remain in the atmosphere for an infinite time ('airborne fraction'; e.g. Joos et al., 2001).
In the current study the impulse–response function (equation eq. 1) described by Maier-Reimer and Hasselmann (1987) was used in a slightly modified version to depict the removal of emitted (anthropogenic) CO_2:

$$\mathrm{f}(t)_{\mathrm{CO_2}} = a_0 + a_1 * e^{(-t/T_1)} + a_2 * e^{(-t/T_2)} + a_3 * e^{(-t/T_3)} \qquad (eq.1)$$

where a_0, a_1, a_2, a_3 describe the share (%) of specific fractions of CO_2 with differentiated lifetimes of T_1, T_2, T_3 (years). The parameters of this function have been taken from Forster et al. (2007). They are based on the revised version of the Bern Carbon cycle model (Bern2.5CC; Joos et al., 2001), and reflect oceanic removal only (Table 1).

Table 1: Lifetimes of the four fractions of emitted CO_2 (according to Forster et al., 2007, based on Joos et al., 2001).

CO_2	**Lifetime (T; years)**	**Percentage (a; %)**
fraction a_0	infinite	21.7
fraction a_1	172.9	25.9
fraction a_2	18.51	33.8
fraction a_3	1.186	18.6

Subsequently, the impulse – response function for the removal of released CO_2 (eq. 1 above) was included into the simulation of atmospheric CO_2 concentration to calculate global emissions occurring over a 250 years time scale (1755 to 2005) for different anthropogenic emission pulses.
For calculation of CO_2-concentration, global annual CO_2 emissions were composed of net (LU)LUC-related CO_2 emissions (i.e. allowing for sequestration in anthropogenic sinks) as given by Houghton (2008; from 1850 to 2005) and other anthropogenic CO_2-emissions (from use of fossil fuels and cement production) as provided by Boden et al. (2009) for the time period from 1750 to 2005. The CO_2 emissions are shown as CO_2-carbon in Figure 1 for the time series from 1850 to the year 2005. Missing data on LUC-related CO_2 for 1755 to 1850 were replaced by extrapolating from the value for the year 1850 (Houghton, 2008).

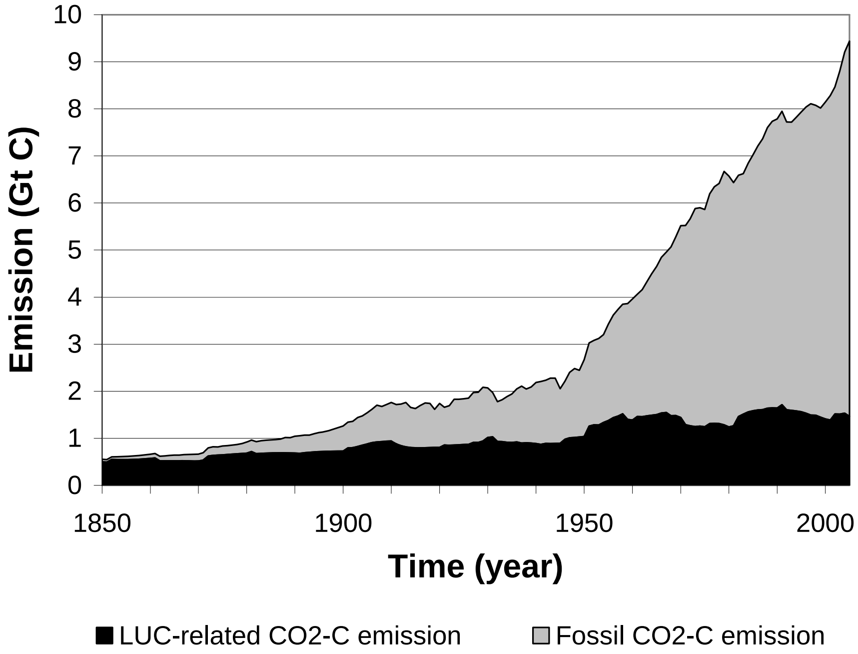

Figure 1: CO_2-C-emissions (Gt C) related to LUC and the use of fossil fuels from 1850 to 2005 according to data from Houghton (2008) and Boden et al. (2009).

The (pre-industrial) 1750 value for the CO_2 concentration of 277.2 ppm (Etheridge et al., 1996) was assumed to have remained in the atmosphere, representing an equilibrium state between natural CO_2-emissions and CO_2-sequestration that had been present for about 10,000 years preceding the industrial period (Denman et al., 2007). Anthropogenic CO_2-emissions occurring since 1755, including LUC were subjected to the removal function as of eq. 1. Any CO_2 not removed was considered to accumulate above the pre-industrial value of 277.2 ppm unless removed by sequestration in terrestrial sinks. This ‘atmosphere-to-land-flux’, was assumed to start at zero (0) Gt C in 1755, to be linearly related to the increase in atmospheric CO_2 concentration above the initial 277.2 ppm and to reach a final value of 3.2 Gt C (calculated based on IPCC-AR4-values for the 1990ies in Denman et al., 2007) in 2005.

2.2 Validation of the simulations of atmospheric CO_2 concentrations

For validation the model output was compared with CO_2-concentrations measured by Etheridge et al. (1996) for the time series from 1755 to 1970 and with values cited by Denman et al. (2007) for the years 1998 and 2005. This comparison showed a mean deviation of 1.8 ppm for the 12 observations as compared to the model predictions (Figure 2). This model to a large extent represents the conception and the parameterisation of the Bern Carbon Cycle Model with parameters given in Forster et al. (2007), thus the validation curve is almost identical to the results presented by Joos et al. (2001).

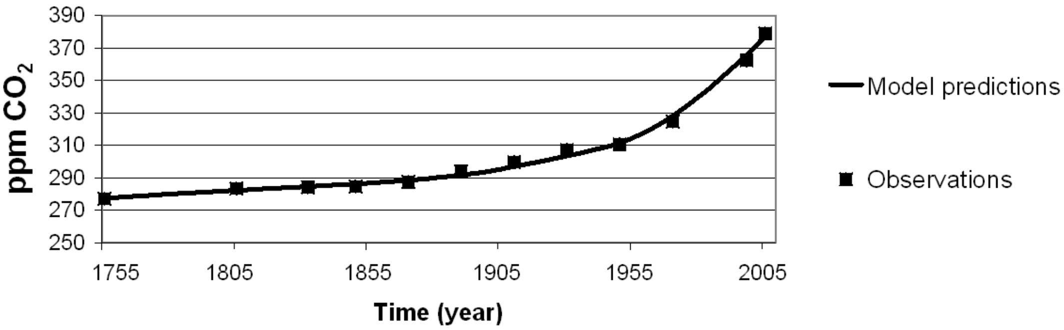

Figure 2: Atmospheric CO_2 concentrations as predicted by the model versus observed values.

3. Results and Discussion

In this chapter results are presented for system boundaries and CO_2 fluxes from LU and LUC which are based on published information (section 3.1). Subsequently, LUC-related emissions and the removal of emitted CO_2 which are derived from model calculation will be presented (section 3.2). Based on this, methods are suggested for the estimation of LULUC-related CO_2, including the effects of different approximations on GHG emissions from LUC (section 3.3).

3.1 Emissions and fate of CO_2 from land use and land use change

This chapter covers the setting of sound system boundaries, the definition of accounting periods, the concept of neutrality of biotic CO_2 and the consideration of potential sinks for GHG.

Fluxes of greenhouse gases within system boundaries

Basically, different approaches are described in the literature for system boundaries, which may include different GHGs in CFs. Based on the literature sources reviewed, we propose a rather broad definition of a CF in accordance with e.g. PAS2050 (2008) and WRI/WBCSD (2009), but in contrast to e.g. Grub and Ellis (2007) and Wiedmann and Minx (2007). Therefore, to our opinion a CF should include at least emissions of CH_4, N_2O (plus chlorofluorocarbon if relevant) and CO_2 as does the GWP in LCAs. Both CFs and GWPs should include not only fossil, but also non-fossil C-emissions from biomass and C_{org}, which have been stored for a long time and will be released as a consequence of LULUC.

Generally, the definition of system boundaries is one of the most important steps in building CFs or LCA-GWPs and their specific setting strongly affects the results. Hence, in estimating GHG emissions for CFs and GWPs, we propose to strictly focus on effective (physical) fluxes of greenhouse gases which occur in connection with the supply chain of a product. Additionally, we propose a setting of system boundaries including processes which are relevant for LUC connected to the production of food and feedstuffs, i.e. where forests savannahs and grasslands are converted into grassland and arable land, respectively. Other categories of land are proposed not to be considered. As an exemption, LUC from arable land to perennial grassland could be considered, as this process sequestrates CO_2 which should be accounted for. Due to high imports of cash crops, especially oil-seeds such as soybean with relevance for European livestock production as opposed to beef imports into Europe (European Commission, 2011), the main focus of this study lies on the conversion of land into arable land. It is important to note that the provision of 1 ha of grassland which originates from LUC from primeval or secondary forests results in similar changes in C_{org} for arable land. As opposed to LUC-related CO_2 fluxes for production of food and feedstuffs, which result in one-directional fluxes to the atmosphere only (emissions), certain forms of LU may also act as a sink for greenhouse gases.

Accounting period for LULUC-related emission

LUC only has to be considered for a specified time after changes have occurred from one land use category to another. This LUC results in CO_2-emissions from biomass and from soil until a new equilibrium state will be reached for C_{org}. In contrast, GHG emissions from LU (due to e.g. crop rotation, fertilizing/manuring, different types of tillage etc.) have to be permanently considered for arable land.

While the flows of CO_2 from soils which effectively occur as a consequence of LU and LUC can be accounted directly as emissions over a specified time span, GHG emissions from burning and decomposition of biomass which has not been removed from the area have to be assessed differently. On the one hand, if CO_2 rapidly released from biomass (or C_{org}) is included in estimations of GHG emissions, the resulting GHG emissions could be accounted for only the first year of agricultural use. On the other hand, the productive period of farmland which originated from LUC is usually greater than one year, probably within the magnitude of the duration which is needed to find a new equilibrium state in C_{org}. Therefore and for reasons of simplicity, we propose to allocate GHG emissions from burnt and cleared biomass to the same time period accounted for CO_2-emissions from soil after LUC.

Houghton and Hackler (2001) defined time periods of 30 and 5 to 20 years, during which CO_2 is emitted as a consequence of LUC in temperate and tropical regions, respectively, until a new equilibrium in soil C_{org} has been established in the newly developed farmland. For the purpose of simplification, a default period of 20 years for all LUC-related emissions from soil may be applied as it is recommended in guidelines of the International Panel on Climate Change (IPCC, 2006; Watson et al., 2001) for C_{org}-fluxes in national greenhouse gas inventories. Furthermore, this default period may also be used for product-related CFs and GWPs unless data are available which better represent the conditions for emission from soil (IPCC, 2006). This aspect is also covered in chapter 3.2 of this paper.

Following a simplified method for LULUC-related emissions which is based on Renewable Fuels Agency (RFA; 2008), average loads of CO_2 released from soil and biomass are calculated for a given product (see PAS2050, 2008; Searchinger et al., 2008; Hörtenhuber et al., 2011). These emission loads are not directly representing the release curves for C_{org}, but are related to the quantities of product harvested during the respective time period. Consequently, this method of linear allocation produces average values despite the shape of the CO_2 emission curves for C_{org} showing a much stronger change in the beginning of the period and then levelling off (see West et al., 2004). With respect to general uncertainties in estimation of LULUC-related emissions and as a consequence of average values for areas converted in different points of time, we consider this acceptable unless emissions or the area of converted land fluctuate strongly over the time period accounted for.

Land Use Change and 'neutrality' of CO_2 from biomass

There is an intense debate among experts dealing with CF and LCA, whether carbon emitted from above-ground biomass and/or soil should be accounted for at all (e.g. Dalgaard et al., 2008). In most previous studies (LU)LUC-related emissions from biomass which is burnt or removed for utilization is neglected. However, 'neutrality' only exists theoretically for emissions from both soil and vegetation (Marland, 2010; Searchinger et al., 2009; Johnson, 2009), because carbon release and its storage in vegetation regrowth do not occur at the same time or in the same place, this setting not being properly represented by spatial and temporal accounting system boundaries (Marland, 2010). 'CO_2-neutrality' was in place only before industrial revolution, when the balanced ratio between carbon emission and CO_2-uptake capacity resulted in a nearly constant atmospheric CO_2 concentration between 260 and 280 ppm over the last 10,000 years before 1750 (Denman et al., 2007).

The high amounts of fossil CO_2 emitted into the atmosphere (Figure 1), together with huge loads of LUC-related CO_2, which are released from certain regions within short periods of time are far beyond the uptake capacity of the vegetation, which is even assumed to decrease (Denman et al., 2007). This is particularly relevant for tropical regions, where carbon stores built up over at least 6,000 years without effects on the atmosphere and other biomes (Foley, 1994). In most other ecosystems, soils are quantitatively more important than vegetation for carbon storage (Watson et al., 2001). Since LUC currently occurs mainly in tropical regions, LULUC-related emissions from both carbon pools have to be considered in CFs and GWPs.

In contrast to LUC, CO_2 from above-ground biomass which is released due to LU (i.e. agricultural production on established agricultural land), is regarded as part of the carbon cycle and hence as CO_2-neutral: Uptake of CO_2 by vegetation and its release after oxidation occur annually or within a few years. Therefore, emissions from LU to be considered in CFs and GWPs were restricted to emissions from soil (variation in C_{org}-content) in this study.

Consequently, this approach also does not take into account a hypothetical source ‘loss of the sink function’ for arable land (meaning that the utilization of arable land prevents its return to grassland or forest which could act as carbon sinks), as advocated by Milà i Canals et al. (2007), Kool et al. (2009) as well as Müller-Wenk and Brandão (2010). The main argument for excluding this hypothetical source is that it is not paralleled by physically occurring carbon fluxes and that its inclusion would be contradictive to the definition of temporal system boundaries which can be directly related to physically occurring carbon fluxes. It should be kept in mind that imposing a ‘loss of the sink function’ for arable land will exaggerate emissions in CFs or GWPs of products originating from land areas which have been converted to agricultural land long ago. Accordingly, carbon emission loads will be underestimated for products from recently converted agricultural land areas, although these may eventually emit even higher CO_2 loads, e.g. in the case of tropical grassland and forests recently converted into agricultural land (Watson et al., 2001). This issue will be followed up in section 3.2.

3.2 Results from simulating atmospheric CO_2 concentration changes

The removal of LULUC-related CO_2

The effect of one unit of emitted CO_2 on the current atmospheric CO_2 concentration depends on the relative point in time at which the emission took place. This, together with the fact that CO_2 emissions rose tremendously during the last 2.5 centuries, led to a substantial increase in atmospheric CO_2 concentrations during the last 100 years. The relative amounts of historically emitted and (LU)LUC-related CO_2 which are still present in the atmosphere, i.e. not removed by oceanic pools or terrestrial sinks, were derived from the simulation of the atmospheric CO_2 concentrations described above and are shown in Figure 3.

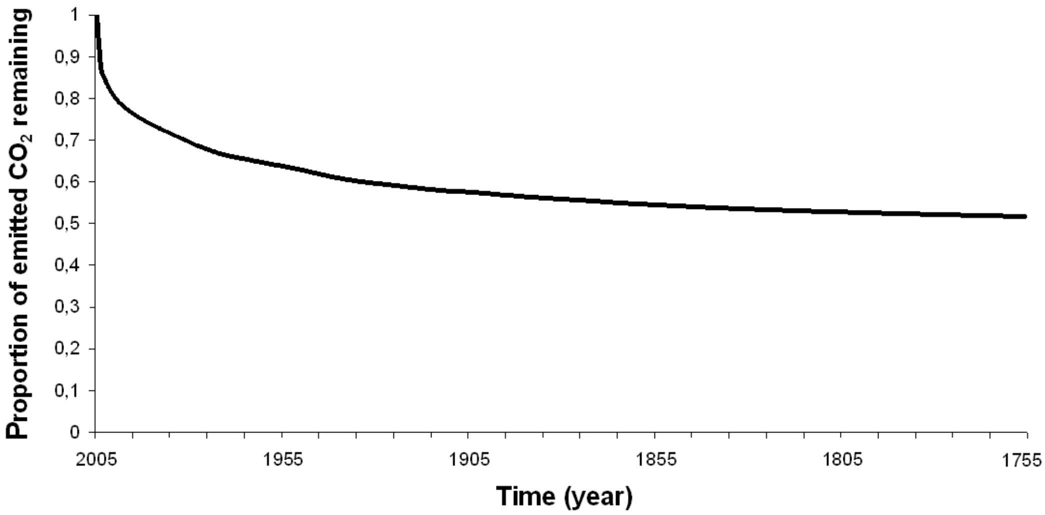

Figure 3: Proportion of emitted LULUC-related CO_2 remaining in the atmosphere integrating over all emissions between a certain year and 2005[1].

Assuming a spike emission of CO_2 into the atmosphere, it will take 31 years until more than 50 % of the released amount are taken up by oceans (i.e. half-life period of 31 years; derived from eq. 1). The situation for the atmospheric fate of the total CO_2 which was emitted globally over the whole time period considered is more complex (see also Figure 3): As time was too short for efficient removal of CO_2 emitted in recent decades, relatively high shares of recent (LU)LUC-related CO_2-emissions still remain in the atmosphere. Additionally, annual CO_2-emissions were not constant over time, but increased dramatically during the last few centuries. This particularly applied to fossil CO_2, but also LUC-related emissions still increased in recent centuries. A large share of these recent emissions is still present in the atmosphere. Consequently, the proportion of LULUC-related CO_2 remaining in the atmosphere is higher than would be expected if deriving from a spike in the beginning of the observation period.

To show effects and time characteristics of LUC on GHG emissions from C_{org} losses for a specific area, Figure 4 takes into account (a) LUC-related flux of CO_2 released from soil to atmosphere based on West et al. (2004; dashed curve) and its fractions which are not removed by (b) oceanic pools (black curve; eq. 1 applied) and (c) oceanic plus terrestrial sinks (grey curve). Although considered separately for an isolated area of converted land, emissions have to be considered

[1] E.g., 68 % of LUC-related CO_2 emitted during the 30 years between 1975 (x-axis) and 2005 are still present in the atmosphere in 2005.

within the total pool of globally emitted CO_2. Hence, a globally occurring removal of CO_2 (see eq. 1) was assumed to affect the released CO_2, with a constant natural flux to terrestrial sinks over the 30 years time period.

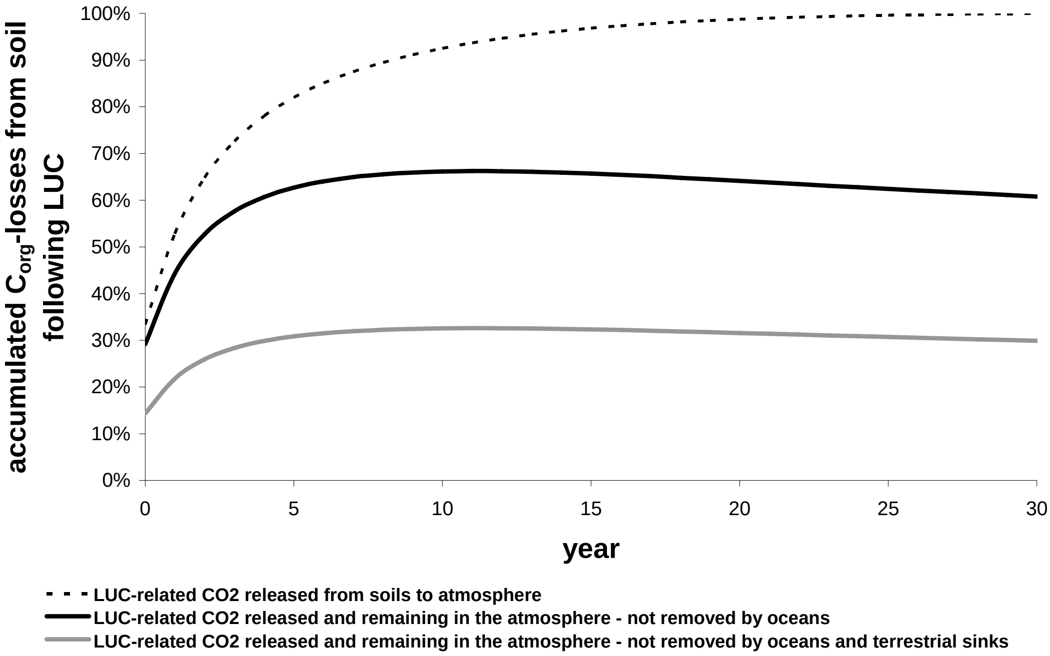

Figure 4: Time characteristics of CO_2 release from soil vs. CO_2 removal from the atmosphere.

Figure 4 shows a decline of emitted CO_2 remaining in the atmosphere after ten to 15 years. These ten to 15 years may serve as a first recommendation for derivation of appropriate accounting periods, although globally applicable periods may be better derived from global historical LULUC-related GHG emissions (Figure 3). This is discussed in detail below (see subchapter 'Appropriate accounting periods').

LULUC-related and fossil CO_2-emissions

According to the input data used herein, 61 % (321 Gt C) and 39 % (204 Gt C) of total CO_2 emitted between 1755 and 2005 can be related to the use of fossil energy sources and to LUC, respectively. Due to a longer time span for the removal of emissions caused by LUC in former times than for more recent CO_2-emissions from the use of fossil sources, about 77 ppm (77 %) and 23 ppm (23 %) of the atmospheric increase of CO_2 between 1755 and 2005 can be allocated to fossil and LUC-related sources, respectively. Carbon cycle simulations by Brovkin et al. (2004) showed highly comparable results, making LUC-CO_2 responsible for 22 to 43 ppm of the total CO_2 increase during the last millennium.

The substantial proportion of roughly 23 % LULUC-related increase in CO_2-concentration emphasizes its contribution to total CO_2-emissions being highly relevant and hence supports its incorporation into CFs and GWPs for food and feedstuffs if these sources actually exist within the respective supply chain. Accordingly, studies which do not account for emissions occurring from LUC and LU for food, feedstuffs or bio-energy underestimate the increase of the CO_2-concentration in the atmosphere by more than 20 %. Therefore, emissions from LULUC should in any case be included in CFs and GWPs for LCAs.

It is important to note that net emissions from LUC (i.e. difference between total CO_2 emissions and CO_2 sequestrated in anthropogenic sinks) were used as input data (Houghton, 2008) for simulation of CO_2 concentrations. This is in contrast to the definition of system boundaries for LUC (see subchapter 'Fluxes of greenhouse gases within system boundaries') which implies that only processes related to an agricultural land use and thus only emission, but not LUC-related sequestration should be included into CFs or GWPs of feedstuffs and food. Obviously, land which is changed from arable land to woodland is not available for food or feed production anymore; therefore the relevance of LUC in the increase of atmospheric CO_2 can be expected to rise remarkably if LUC input data is changed to gross emissions, i.e. without reflecting the CO_2 sequestrated in terrestrial anthropogenic sinks. , for application of methods presented in chapter 3.3, LUC-related GHG emissions reflect gross values.

The airborne fraction of CO_2 and its cut-off

One fraction of emitted CO_2, the so-called 'airborne fraction' (21.7 %) is assumed to remain in the atmosphere for an infinite time (Table 1; Joos et al., 2001). As this fraction is not reflected in all previous studies concerning effects on LULUC, some published results may be misleading. A full consideration of the airborne fraction and application of eq. 1 result in estimates for the atmospheric CO_2 concentration which is in accordance with measured data (see chapter 2, validation). Contrarily, deviating atmospheric CO_2 concentrations result from assuming a linear removal of CO_2 including an average residence time in the atmosphere and a cut-off for CO_2 as is suggested by Müller-Wenk and Brandão (2010). For example, a 500 year cut-off for the airborne fraction, as reflected in the latter study, implies that all emissions would be finally stored in the terrestrial biosphere and in oceanic carbon pools after 500 years. This would require an average linear removal over a period of 157 years, leading to a clear overestimation of the current CO_2 concentration (439 ppm). This is mainly due to the relatively slow mean linear removal rate while the initial fast removal of any concentration dependent algorithm is ignored. Contrarily, a 100 year cut-off as suggested by Müller-Wenk and Brandão (2010) with a linear removal over 47.5 years underestimates current CO_2 concentration (359 ppm) due to a relatively high linear mean removal as compared to the concentration dependent curve.

Appropriate accounting periods for emission and removal of LULUC-related CO_2

As a consequence of the removal process, the proportion of emitted CO_2 remaining in the atmosphere is inversely related to the observation period (Figure 3): While only 57.7 % of all LUC-related, global CO_2 emissions of the last 100 years (1906-2005) are still in the atmosphere, a higher value of 71.4 % CO_2 would result from defining an observation period of 20 years (1986-2005). Consequently, long observation periods, e.g. 250 years (1756-2005) which are sometimes chosen in order to include more relevant phases of anthropogenic LUCs (e.g. Kool et al., 2009), would result in only 51.7 % of emitted CO_2 still remaining in the atmosphere. Hence, almost 50 % of the CO_2 released over the last 250 years would have been removed from atmosphere until today and would not contribute to global warming anymore. Long accounting periods (i.e. 30 years and longer) likely also cause the problem of lacking accurate historical data concerning LULUC-related emissions, area of converted land, areas of cultivated crops and related yields, etc. Additionally, the relevance of changed production patterns (e.g. introduction of new crops, dropping of other crops) are not easy to account for.

On the contrary, very short accounting periods which would show a high proportion of emitted CO_2 still remaining in the atmosphere, do not cope with a first recommendation for allocating emissions over the specified time span they are emitted. Short accounting periods of e.g. 1 or 5 years would result in 87.5 % and 80.0 %, respectively, of emissions from this period still remaining in the atmosphere, but they would not include all relevant emissions as C_{org} is emitted for up to 30 years after LUC, especially for temperate soils (Houghton and Hackler, 2001). However, due to the logarithmic shape of the CO_2 emission curves (West et al., 2004) about 93 % and 99 % of emissions from C_{org} are released within 10 years from temperate and tropical soils, respectively. Consequently, besides the 20 years accounting period as derived from IPCC-guidelines (IPCC, 2006) a shorter default accounting period of 10 years could be applied, which would include the vast majority of LUC-related soil emissions and roughly three quarters of the emissions remaining in the atmosphere.

3.3 Methods for estimation of emissions from LULUC and their effects on emission loads of food and feedstuffs

Emissions from Land Use Change (LUC)

Equation 2 and the following Table 2 describe a simple allocation of LUC-related emissions to the respective products. The case of Brazilian soybean production is used as an example for LUC, which occurred due to the growing demand for agricultural area for production of food, feed and agro-fuels. Brazil ranks first in global exportation of several agricultural products such as sugarcane, beef and chicken, it is second in exporting soybeans and maize (Brazilian Ministry of Science and Technology; BMST, 2010) and is considered one of the world's top GHG emitting countries (Cerri et al., 2009).

$$\mathrm{LUC\_EM\_crop}_{n,c} = \mathrm{LUC\_EM\_tot}_{n} * \mathrm{LUC\_F}_{n,c} / (\mathrm{area}_{n,c} * t * Y_{n,c}) \qquad (eq.2)$$

where $LUC\_EM\_crop_{n,c}$ is the LUC-related emissions per kg DM (dry matter) of a specific crop c for nation n, LUC_EM_tot characterises the total national LUC-related GHG emissions (excluding sinks, i.e. without subtracting of sequestrated GHG) for nation n during the accounting time period (t). The LUC-related factor $LUC\_F_{n,c}$ represents the proportion of newly converted land that is occupied by crop c for nation n. The area of arable land cultivated with crop c for nation n is included as $area_{n,c}$. $Y_{n,c}$ denotes the DM-yield per ha for nation n and crop c. For the accounting time period t default values of 10 or 20 years are used herein (see section 3.2 above).

Information on LUC_EM_tot is available from national GHG-Inventories or national communications to the United Nations Framework Convention on Climate Change, sometimes also from relevant research papers. If emissions were to be estimated more accurately, $LUC\text{-}EM_{tot\ n}$ could be calculated from values for the converted land area and their shares in the original land use category (i.e. primeval or secondary forest, savannah or grassland) and emission data from relevant sources, e.g. according to the PAS2050-guideline (RFA, 2008) or Don et al. (2010). The calculation of the LUC-related factor $LUC\_F_{n,c}$ for a crop has to be based on the total land area that is converted into agricultural land, including converted pastures and grassland for feeding livestock. It is important to calculate LUC-related CO_2 according to the best available data, eventually allowing for a mass-flow approach (i.e. which type of former land is converted into which type of land for agricultural production including individual C_{org}-contents before and after LUC).

In Table 2 a simple estimation is used to illustrate eq. 2, considering total amounts of CO_2 released from LUC in Brazil for two different accounting periods, demonstrating a trend for the case of soybeans which reflects the decreasing relevance of soybean production. Hence, the LUC-factor decreases with the 10-years accounting period and consequently the emission of LUC-related CO_2 per kg of feedstuff is reduced. The missing proportion of emissions from LUC which is not related to crop c (e.g. soybean: 76 % and 82 % for the 20- and the 10-year accounting period, respectively) has to be allocated to pasture or other crops, e.g. sugar cane, which showed an increase during recent years (Food and Agricultural Organisation of the United Nations; FAO, 2010).

Table 2: Estimation of LUC-related emissions for Brazilian soybean (kg CO_2-eq kg^{-1}) for different accounting periods.

	Accounting period in years (t in a)	National LUC-related emissions for relevant agricultural land over the accounting period ($LUC\_EM_{tot\ n}$ in kg CO_2-eq)	Fraction of national LUC-related area for crop c ($LUC\_F_{n,c}$)	Agricultural land cultivated with crop c for nation n ($area_{n,c}$ in ha)	Average DM-yield per ha ($Y_{n,c}$ in kg)	Result (kg CO_2-eq)
Data sources	see section 3.2	National GHG-Inventories, similar communi-cations, research papers (Cerri et al., 2009)	FAO (2010)	FAO (2010)	FAO (2010)	
Brazilian soybean	20	20.6*10^12	0.24	15,247,333	2,403	6.740
Brazilian soybean	10	10.7*10^12	0.18	19,242,252	2,566	5.169

Despite differences in the approaches used, other studies report similar results for LUC-related emissions: 4.15 and 5.41 kg CO_2-eq kg^{-1} DM were found for extracted soybean meal originating from Brazil (78 %) and Argentina (20 %) and for South American extracted soybean meal which was a co-product from the production of agro-fuel (Hörtenhuber et al., 2011; Hörtenhuber et al., 2010a), respectively. Deducting the effects of extraction (i.e. co-product oil), LUC-related emissions of 6.08 and 6.18 kg CO_2-eq can be expected for 1 kg DM of unprocessed soybean according to Hörtenhuber et al. (2011) and Hörtenhuber et al. (2010a), respectively. Similarly, Dalgaard et al. (2008) estimated GHG emissions from LUC (above-ground biomass) to be about 5 kg CO_2-eq kg^{-1} soybean.

Because of the emerging relevance of the production of agro-energy, a method for the estimation of 'indirect LUC' (iLUC) has been developed by a number of authors (e.g. Plevin et al., 2010; Cherubini et al., 2009) and is currently extensively discussed in the literature. iLUC represents a theoretical LUC which occurs elsewhere on the globe due to the production of the agricultural goods which have been outcompeted by the (agro-energy) crop in question. Despite the arguments which could be brought forward for an inclusion of iLUC into LUC models, it is not covered herein, mainly because of the difficulties of relating it to physically occurring carbon fluxes. The difficulties coming along with the inclusion of iLUC are discussed by other authors (Plevin et al., 2010).

Emissions from Land Use (LU)

Emissions from land-use (LU; see equation 3, Table 3) can be calculated following the same approach as for LUC (equation 2, Table 2):

$$\mathrm{LU\_EM\ kg^{-1}\ crop_{n,c}} = \mathrm{LU\_EM\_tot_{n}} * \mathrm{LU\_F_{n,c}} / (\mathrm{area_{n,c}} * t * Y_{n,c}) \qquad (eq.3)$$

where $LU\_EM\_tot_n$ and $LU\_F_{n,c}$ characterise the LU-related emission and its factor, describing the effects of a change in C_{org} stocks due to agricultural land use. For the purpose of simplification, we will here express the land use factor $LU\_F_{n,c}$ as the proportion of crop c from total agricultural land for nation n.

Table 3: Estimation of LU-related emissions for Brazilian soybean (CO_2-eq kg^{-1}) for different accounting periods.

	Accounting period in years (t, years)	National LU-related emissions for relevant agricultural land over the accounting period ($LU\_EM_{tot\ n}$, kg CO_2-eq)	Fraction of national LU-related area for crop c ($LU\_F_{n,c}$)	Agricultural land cultivated with crop c for nation n ($area_{n,c}$, ha)	Average DM-yield per ha ($Y_{n,c}$, kg)	Result (kg CO_2-eq)
Brazilian Soybean	20	1.48*10^12	0.059	15,247,333	2,403	0.1189
Brazilian Soybean	10	0.66*10^12	0.073	19,242,252	2,566	0.0976

As compared to Central European conditions for conventional feed production (Hörtenhuber et al., 2010a), LU-related emissions from arable land seem to be remarkably higher (factor of 2) for important South American production areas. Different management practices are supposed to be one of the reasons for this difference. These include crop rotations or the application of manure which is more frequently practiced in mixed farming systems and which supports more balanced C_{org} contents. Still, GHG emissions from LU are much smaller than from LUC (see Tables 2 and 3).

Practical relevance of LULUC-related emissions for the overall emission burden of food and feed

For specific crops such as soybeans, consideration of (LU)LUC-emissions have a high impact on the results, as all other sources of GHG emissions (soil-N_2O, use of fuels and energy for agricultural activities, transports, industrial, mineral fertiliser and pesticide production) have only a relatively small influence. In total, the latter factors contribute less than 1 kg CO_2-eq kg^{-1} DM, despite the long transport distance between South America and Europe (Hörtenhuber et al., 2011), as compared to the 5 to 6 kg CO_2-eq kg^{-1} DM from (LU)LUC for non-certified soybeans (i.e. not certified according to the 'Basel criteria'; ProForest, 2004). Subsequently, GHG emissions from (LU)LUC play an important role for CFs and GWPs of all agricultural products which have been produced by utilizing substantial amounts of (LU)LUC-burdened means of production, such as extracted soybean meal. This is particularly relevant for conventionally produced pork and poultry meat, where replacement of extracted soybean meal is not as easy as for ruminants (Hörtenhuber et al., 2011). First calculations in which the LUC-related GHG emissions for extracted soybean meal (as derived by Hörtenhuber et al., 2010a) were used to estimate total GHG emissions from conventional and organic broiler chicken production (Hörtenhuber et al., 2010b) point to about 50 % lower emissions from the organic production system, in which no extracted soybean meal is used. This difference is entirely related to LUC. This source is less dominant for milk production (up to 9 % of total emissions per kg milk from a grassland-based production system; Hörtenhuber et al., 2010a), but may also play a role there, especially in high-output production systems if those partially rely on imported soybean meal or other externally produced feedstuffs (Hörtenhuber et al., 2011).

4. Conclusion and Perspective

From the literature review and the model calculations presented herein, it is concluded that system boundaries should be defined rather broadly in the estimation of carbon footprints and global warming potential of agricultural production. This means that at least CH_4, N_2O and CO_2 from fossil sources and the degradation of above ground-biomass plus C_{org} should be taken into account. It is suggested that GHG emissions accounted for in carbon footprints and global warming potential should be restricted to physically occurring fluxes of greenhouse gases and should exclude hypothetical sources such as a 'loss of the sink function' for arable land. CO_2-neutrality for emissions from LUC only exists theoretically as the storage of released carbon occurs over substantial time and not necessarily in spatial proximity.

While different accounting periods may be appropriate for different purposes, also depending on regional conditions, 10 or 20 years may both be used as a suitable general default period. The 10 year accounting period includes the majority of LUC-related emissions released from soils as well as of emissions remaining in the atmosphere. The 20 year-period may serve as default in between the 30 and 5 to 20 years during which CO_2 is emitted as a consequence of LUC in temperate and tropical regions, respectively, and is more feasible for temperate conditions with their lower rates of CO_2-release. However, for specific (e.g. product-related) CFs and GWPs, specific accounting periods should be defined, depending on the time scale for physically occurring LUC-emissions.

Emissions from LUC contributed 23 % to the increase in atmospheric CO_2 concentration of the last 250 years. Therefore this important source for emissions needs to be included in the estimation of CF and GWP, where great quantities of LULUC-burdened inputs, such as extracted soybean meal, are used. This can be illustrated by an estimated 50 percent less CO_2-eq per kg of broiler carcass if LUC-loaded feedstuffs (particularly extracted soybean meal) are substituted for.

The implementation of the methodological approach for the coverage of LULUC in the estimation of GHG emissions which are discussed herein offers a practicable tool for the assessment of CFs and GWPs of agricultural supply chains. Depending on the geographical area to be addressed in such an assessment, the availability of sound data for areas in which LUC occurred may be a critical factor.

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3 Overall discussion and Conclusions

This chapter contains a condensed, but detailed discussion of GHGE from Austrian dairy PS in the light of most recent publications and of a specific mitigation strategy for certain PS. Furthermore, the results are related to different assessment approaches, with a specific focus on land use and land use change.

3.1 GHGE from Austrian milk production systems and a comparison with results from recent literature

Two studies on GHGEs from dairy production systems were found which are relevant for a comparison of results and methods of the first publication in this thesis (Gerber et al., 2010; Leip et al., 2010). Both studies were published after publication 1 (Hörtenhuber et al., 2010; see section 2.1) and were hence not yet reflected in it, but showed comparable methods, system boundaries and results. Additionally, GHGE per kg raw milk (results) for Austria presented in publication 1 (section 2.1) can be compared with those for other countries, on a global and continental scale with FAO-results (Gerber et al., 2010), and on an European scale with Leip et al. (2010).

According to this comparison, raw milk from eight typical Austrian production systems (PS) shows very low GHGE per kg milk of between 0.81 and 1.17 kg CO_2-eq (section 2.1, Table 7, Fig. 1). Based on this and on data for the regional distribution of milk production in Austria (Kirner, 2009), the national average is to be estimated at 1.04 kg CO_2-eq per kg raw milk. This is in agreement with the average 1.0 kg CO_2-eq reported for Austria by Leip et al. (2010). This source even gives an estimate of only 0.7 kg CO_2-eq per kg raw milk for the Austrian province of Tyrol. Both values are presented as the lowest among comparable spatial scales (countries and regions, respectively). This is explained as a consequence of low GHGE from land use and land use change (LULUC), indicating the use of high proportions of forage and home-grown roughage in the diets of the dairy cows.

Contrarily, alpine PS in regions such as Tyrol, which are characterised by high proportions of forage in the diets and therefore by relatively low milk yields, are expected to show high methane emissions from enteric fermentation. This is also reflected in publication 1 (section 2.1, Table 2, Fig. 7), in which equal (for organic milk) or even higher GHGE (for conventional milk) per kg raw milk are reported for alpine PS as compared to the Austrian average. These differences can be explained by the different approaches used: Leip et al. (2010) applied the method according to IPCC (2006) for estimating enteric fermentation, which only considers gross energy intake and digestibility of the diet. In contrast to this, the method published by Kirchgessner et al. (1995) was used in publication 1 (section 2.1, Material and Methods, 'Sources of emissions') and hence allows for a more detailed consideration of dietary composition. However, one disadvantage of this approach may be a slight overestimation of enteric methane emissions (see section 2.1, Results and Discussion, 'GHGE from dairy PS').

Regardless the methodology which was used in different studies, average Austrian milk production shows low emissions if compared at a global scale (Gerber et al., 2010): average emission levels per kg raw milk are about 1.2 kg and 1.3 CO_2-eq for Western Europe excluding and including LULUC-related emissions, respectively. Lower GHGE were found for average North American milk production (1.0 kg CO_2-eq), slightly higher GHGE for Oceania, Eastern Europe and the Russian Federation).

The highest (LU)LUC-related GHGE per kg of raw milk are reported for Western Europe (Gerber et al., 2010). This is mainly due to soybeans and soybean meal imported from Brazil and Argentina; soybeans produced elsewhere and other crops were not associated with land use change. Although differing emission factors were used by different authors for soybeans and their cakes or solvent-extracted meals, the LUC-related loads per kg of raw milk are similar for Western Europe (9 % of total CO_2-eq from LUC; Gerber et al. 2010) and the Austrian average (7 %; calculated based on section 2.1, Table 7, and Kirner, 2009). For Austrian milk production, Leip et al. (2010) indicated 5 % of total GHGE originating from LUC and another 2 % from LU. A respective proportion of LU-related GHGE was calculated based on section 2.1, Table 7, and Kirner (2009) and was estimated at 1 % of total emissions. Gerber et al. (2010) excluded LU-related GHGE from system boundaries for calculation.

3.2 Substitution of feedstuffs as a mitigation strategy: a synopsis from publication 1 and publication 2

Inclusion of LULUC-related GHGE into system boundaries results in higher absolute emission values in case of utilisation of LULUC-burdened feedstuffs on the one hand, but introduces options for mitigation as well as for offering the benefits of utilisation of GHGE-friendly feedstuffs on the other hand. Publication 2 (Hörtenhuber et al., 2011a; section 2.2) especially emphasised the advantages of alternative protein-rich feedstuffs (APRFs) as opposed to the use of solvent-extracted soybean meal (SBME) in terms of GHGE. Therein, substitution of SBME in dairy cow diets by 12 nutritionally equivalent mixtures of alternative protein-rich feedstuffs (APRMs) resulted in a reduction of total GHGE of 42 % on average (22 - 62 %). When all other sources for GHGEs connected to dairy production are considered (i.e. enteric methane emissions, manure management systems or use of fossil energy, etc.; see section 2.1), the GHGE mitigation potential of APRMs is on average 23 % (16 - 26 %; see Fig. 1).

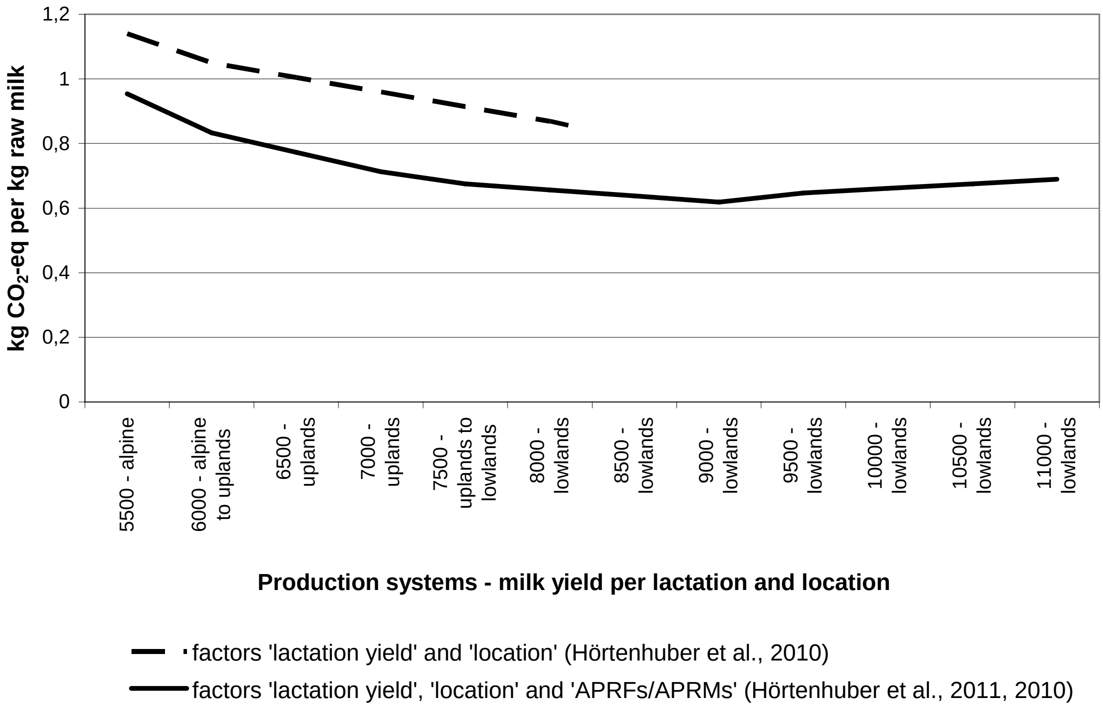

Figure 1. GHGE (kg CO_2-eq) per kg Austrian conventional milk, taking into account effects of different location of production, milk yield and protein-rich feedstuffs.

Figure 1 describes GHGE per kg of conventionally produced Austrian milk depending on the factors *milk yield per cow and year* as well as the *location of the modelled production system* (dashed line). This curve is derived from publication 1 (section 2.1, Table 7) and is based on identical mixtures of concentrates rich in protein for every production system and hence for varying milk yields. Protein-rich concentrates were assumed to consist of one third SBME, rapeseed cake and faba beans each. The full line also represents values based on data for GHGE from publication 1 (section 2.1, Table 7), but accounts for a replacement of the protein-rich concentrates by APRFs and APRMs as described in publication 2 (section 2.2, Table 2). For a milk yield of 5,500 kg per cow and year the protein-rich concentrate was assumed to consist of faba bean only, which was found to be sufficient for supplementing typical Austrian forage-based diets at this relatively low milk yield. As the nutritional value of protein feedstuffs (including the content of rumen undegradable protein) has to increase with rising milk yields, faba beans were assumed to be increasingly substituted by APRMs 7-12 (see section 2.2, Table 2) for medium milk yields. APRMs 1-6 were used for higher milk yield. Finally, a milk yield of 11,000 kg per cow and year made it necessary to include high dietary proportions of SBME and only a low percentage of APRFs, as SBME at the same time contain high protein and amino acid contents, low levels of anti-nutritive constituents as well as a high energy content. The full line therefore shows the lowest GHGE at a milk yield of 9,000 kg per cow and year; a further increase of milk yield results in increasing GHGE per kg milk, mainly due to the high GHGE from LUC connected to the use of SBME.

It has to be noted that the mitigating effects described above are valid only for the respective production system (PS; i.e. location, dietary composition, milk yield level etc.) described in the model calculations. Consequently, statements or recommendations for GHGE-mitigation always have to be related to the specific PS. For example, if relatively high amounts of concentrates are imported into an alpine PS in order to increase milk yield, its mitigation effect per kg milk may very well be lower as compared to a lowland PS in which the same amount of concentrates is used. Generally, feeding of concentrates has advantages in terms of overall GHGE as a consequence of lowering enteric methane emissions and at the same time increasing milk yield (see below for an increase in milk yield from 6,500 to 7,500 kg). However, this is primarily the case if they are produced on-farm, utilizing organic fertilisers and biological nitrogen fixation. If concentrates are not cultivated on farm with adequate yields, relatively high GHGE per kg of feedstuff originating from cultivation (mainly from soil and use of fuels) as well as from long transports worsen the concentrates' GHGE-load. If concentrates are furthermore produced in regions with low stocking rates of livestock and hence with low amounts of manure and only a small proportion of legumes in crop rotations, the use of mineral (nitrogen-) fertilisers fully devours the benefits of lower enteric methane emissions. Additionally, high surpluses of nitrogen in alpine grassland PS with imports of high amounts of concentrates would result in high N_2O-emissions from farmland. Accordingly, milk yields should be linked to the quality of roughage available in the respective region. Furthermore, attempts should be made to increase the utilization of pastures and to improve lifetime performance (see section 2.1, Results and Discussion, 'Mitigation options for dairy PS'). Contrarily, an intensification of Austrian alpine milk production would lead to a net effect of rising GHGE-loads for average Austrian raw milk, and an even worse effect for milk from alpine regions: The attempt to rise milk production per cow and year, for example, from 6,500 kg (i.e. slightly above the Austrian average) to 7,500 kg milk from an equal area of farm land, additional imports of concentrates would be needed. As a consequence, GHGE per kg of Austrian raw milk would increase by about 0.5 % instead of a reduction (calculations based on publication 1. This effect of rising GHGE is due to the concentrates being loaded with high GHGE, mainly as a consequence of LUC. Their import into the PS would counteract the decreasing emissions from enteric fermentation and the "dilution effect" of higher yields.

3.3 Carbon footprints, national inventory reports and their system boundaries

All previous results concerning dairy production which are presented in this thesis reflect the agricultural production of raw milk only, including the inputs necessary for it (see upper box in Fig. 2). Lindenthal et al. (2010) partially based their estimations of GHGE per kg of milk products on methods and results from publication 1, but included the further processing of milk in dairies, storage and transport until the retailer level (Fig. 2).

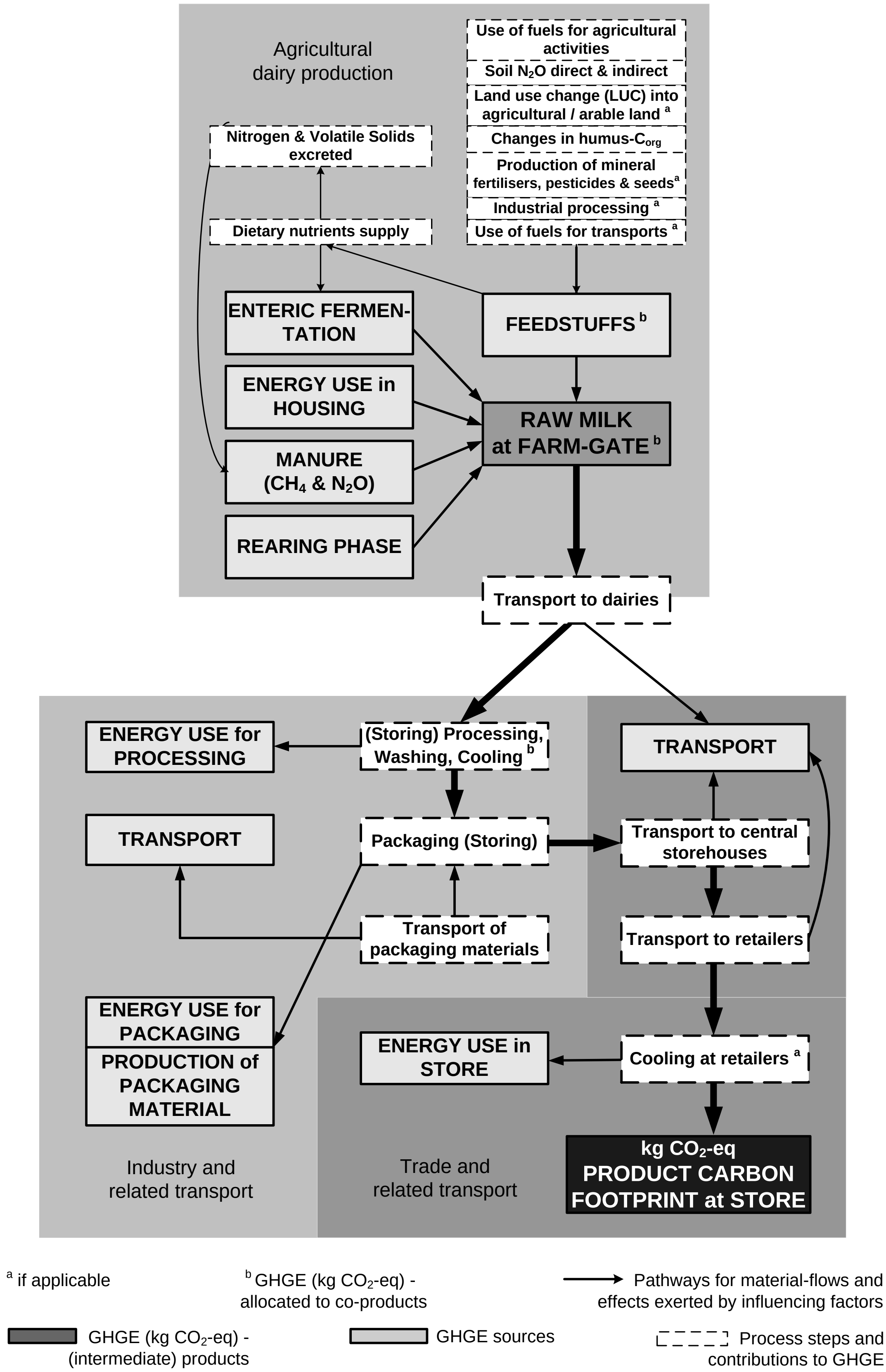

Figure 2. System boundaries for calculation of GHGE for milk (dairy products) at the retailer (own illustration; see also Lindenthal et al., 2010).

Lindenthal et al. (2010) report about 1.2 kg CO_2-eq per kg fresh milk at the retailer level, with post-farm gate emissions accounting for 0.15 kg CO_2-eq per kg fresh milk. If the effects of further processing are included, results for processed Austrian milk are again similar to estimates published by Gerber et al. (2010): assuming the same system boundaries, processed milk is loaded with on average 1.4 to 1.5 kg CO_2-eq per kg in Western Europe. This estimation considers a mix of fresh milk, fermented milk, cream, butter, cheese, whey and milk powder, with post-farm gate emissions varying from 0.06 to 0.23 kg CO_2-eq per kg processed milk.

For the agricultural sector, the National GHGE Inventory Report (NIR; Anderl et al., 2010), which is based on methods revised by Amon and Hörtenhuber (2009, 2010), includes GHGE from soil, enteric fermentation and manure management only. In contrast, ‘carbon footprints’ (CFs) which are comparable to the indicator global warming potential (GWP) in life cycle assessments (LCAs; see publication 3, section 2.3, Introduction), represent overall GHGE occurring within a product’s life cycle or its production phase. As a consequence, CFs or GWPs include GHGE from LULUC as well as from energy supply or transports for both industrial and agricultural processes which are reported in other sectors in NIRs. A product-based view is for example used in Lindenthal et al. (2010) for fresh milk at the retailer level and in publication 1 (section 2.1) for raw milk at the farm gate. Furthermore, differently detailed methodologies may have been used for calculation of specific sources of GHGE in NIRs for some aspects such as GHGE from enteric fermentation. In many cases a less detailed procedure is used for NIRs (mainly ‘tier 1’ or ‘tier 2’ according to IPCC guidelines; e.g. IPCC, 2006), as compared to products’ CFs. While NIRs present results on a national average scale, CFs often reflect other levels of aggregation, allow for regional differentiation or certain specific farm-management factors. Taking into account more specific system elements within CFs involve the use of more precise methods. Therefore and as a consequence of the sectoral view described above, results from NIRs can hardly be compared to results from CFs.

In addition, due to the sectoral perspective, in NIRs even an allocation of GHGE from soil, which result from cultivation of feedstuffs cannot be made to the respective category of livestock. Hence, a NIR does not yield overall results for

GHGE per kg product, nevertheless certain methods (e.g GHGE from manure management systems or from enteric fermentation) used in NIRs are applicable and suitable for the estimation of CFs.

The issue of ‘allocation’ for dairy products mentioned above provides one of the most intensely discussed topics in the scientific community dealing with CFs and LCAs. Additionally, allocation is not only relevant in connection with milk processing, but also with the production of raw milk. No compulsory commitment is formulated for a specific type of allocation (caloric, economic, fat- or mass-related) in the respective guidelines (e.g. BSI, 2008; ISO-14040, ISO, 2006a; ISO-14044, ISO, 2006b). Calculations in both publication 1 and publication 2 were based on a caloric allocation (i.e. according to the energy content of (by-)products), whenever GHGE had to be allocated to two or more (by-) products; this is for example the case for oils and cakes or solvent-extracted meals from oilseeds and for bio-ethanol and distillers dried grains with solubles (DDGS). The case of solvent-extracted soybean meal (SBME) which is typically fed to livestock in Austria demonstrates the consequences of the use of different allocations methods: GHGE per kg DM of SBME vary between 3.85 kg CO_2-eq (80 % of GHGE of the primary soybeans allocated to SBME), 3.28 kg CO_2-eq (67 %) and 2.64 kg CO_2-eq (53 %) if mass, caloric and economic allocation, respectively, is used (publication 2, section 2.2, and own calculations based on it). However, for the Austrian milk production systems, which are covered in this thesis, the choice of allocation methods is without substantial effect on overall GHGE per kg raw milk. This is due to the fact that the diets of dairy cows consist of large proportion of roughages and of only relatively small amounts of by-products from oilseeds or DDGS.

The effects of different allocation types become remarkably greater for processed dairy products, as can be demonstrated for butter: If GHGE are allocated according to fat quantity and without taking by-products such as milk protein and lactose into account (as suggested by e.g. Wiegmann et al., 2005), total GHGE for 20 kg raw milk (i.e. equivalent to an average 20.8 kg CO_2-eq for Austria) would be allocated to one kg of butter. Taking by-products (i.e. skim milk and butter milk) into account and allocating the GHGE load of raw milk according to quantitative proportions, energy quantity or economic value, GHGE per kg butter would strongly decrease (Fig. 3).

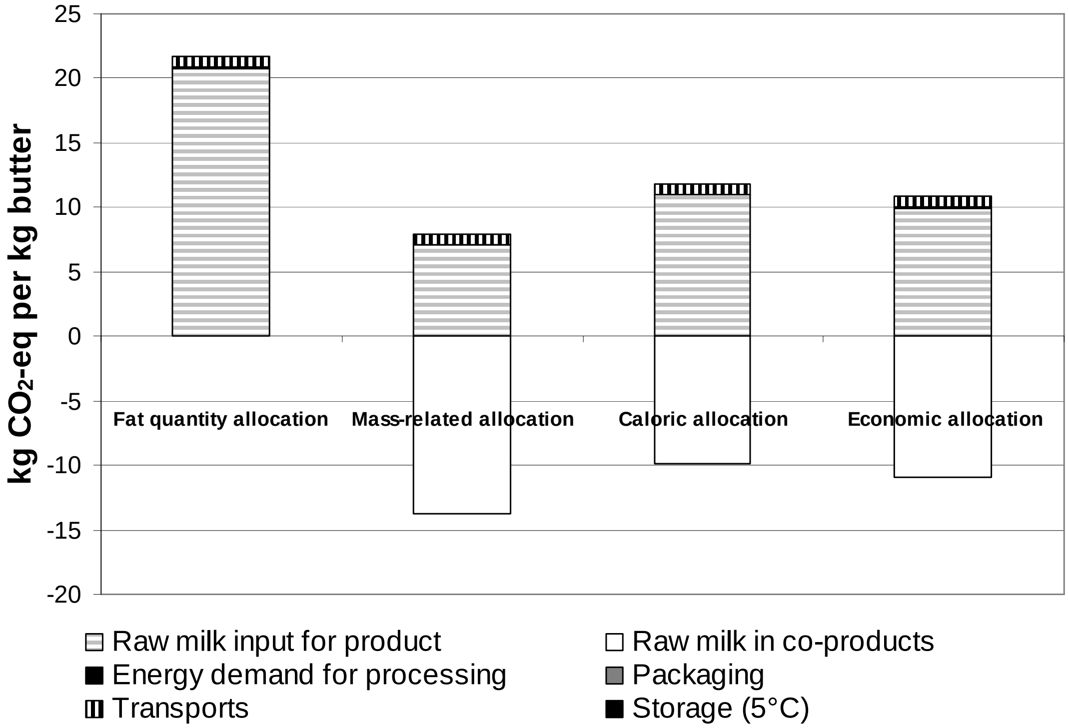

Figure 3. Effects of different types of allocation on the GHGE for butter.

Per kg of product, dairy products high in fat, such as butter, are frequently expected to be loaded with extraordinarily high GHGE, which may eventually be even higher than those of meat products. However, margarine which is produced from oils of imported soybeans and rapeseed may show even higher overall GHGE (11.3 kg CO_2-eq per kg margarine as compared to an average 10.8 kg CO_2-eq per kg butter) if GHGE from LUC, mainly occurring in Brazil, are economically accounted for (calculations based on Fehrenbach et al., 2008, and publication 2). This shows that the tremendous effect of LUC on GHGE is not only related to livestock products and imported feedstuffs, but also to (imported) vegetable food products.

Besides different types of allocation (e.g. economical) which may show relatively lower GHGE for milk and milk products as compared to other allocation methods and other foodstuffs, GHGE of milk production could eventually be allocated to ingredients relevant for milk quality, such as essential nutrients. This may result in specific advantages for milk over other foodstuffs. Furthermore, milk (and beef) production systems may have the unique advantage that they can function solely on the basis of utilisation of plants, which are not directly accessible for human nutrition (i.e. non edible feedstuffs rich in fibre such as forages from grassland). Therefore, within or in addition to future LCAs and carbon footprints (CFs), information should be considered on the efficiency of the respective production system. Approaches such as described by Oltjen and Becket (1996) as well as Gill et al. (2010), who expressed the proportion of 'humanly edible returns' relative to 'humanly edible inputs' may help to assess another important aspect of sustainability; approaches like this which better characterise the contribution of livestock production systems to food security, can be expected to show specific advantages for ruminants as compared to monogastric livestock production systems.

3.4 Effects of different methods for estimation of LULUC-related GHGE

Brazil is the world's largest exporting country of beef and high proportions of newly (i.e. during the last 20 years) deforested land are occupied by pastures (publication 3, section 2.3; Hörtenhuber et al., 2011b), the majority of the national LUC-related GHGE can be directly related to beef production (Cederberg et al., 2011). Nevertheless, 18 % and 24 % of the land area which was deforested during the last 10 and 20 years, respectively, can be directly related to soy production and hence to protein feed export (publication 3). Oil-seeds and especially SBME, but not beef or other meat products account for the greatest proportion of (LU)LUC-burdened goods which are imported to Europe from Brazil. This is mainly due to the gap in self-sufficiency for protein feedstuffs and for the case of Austria additionally due to the self-sufficiency for beef being typically far beyond 100 %. Generally, oilseeds and grains are imported into EU-27 to a much greater extent (in the magnitude of 20 to 30 times) as compared to meat and meat products (European Commission, 2011).

Although gaseous and liquid losses of nitrogen, such as ammonia, nitric oxides or nitrate are not explicitly covered in this thesis these N-fluxes were calculated for an estimation of indirect N_2O-losses (publication 1 and publication 2), according to IPCC approaches (IPCC, 2006) as well as the Austrian NIR (Anderl et al., 2010) and two previous studies for the agricultural sector by Amon and Hörtenhuber (2009, 2010).

Negative anthropogenic effects on ecosystems by nitrogen, phosphorus and CO_2 with the consequences of eutrophication and acidification of aquatic systems (oceans for CO_2) are not reflected in this thesis (publication 1 and publication 2), even though from a mass flow-point of view, they are all highly important. This is the same for carbon which emitted in huge amounts as compared to other nutrients. However, agreements on the 'neutrality' of a high proportion of CO_2 in the organic cycles (sequestration in terrestrial sinks) which IPCC and similar organisations assume, reduce the estimated impacts of CO_2. Nevertheless, CO_2 emissions from burning of fossil fuels and LULUC still provide the highest effect on global warming (IPCC, 2007; publication 3).

If LULUC-related GHGE were not accounted for in publication 1, differences between organically and conventionally produced milk would have been minimised and would better agree with most results of previous studies (which similarly did not account for GHGE from LULUC).

Any scenario which does not include LULUC within its system boundaries, in my view does not reflect reality (i.e. physical CO_2-fluxes to atmosphere which cause global warming). This also applies to a method introduced to LCAs, where LULUC-related GHGE include on the one hand high LU-CO_2 and on the other hand low LUC-CO_2. This is due to the application of a 'loss of sink function' (see publication 3 and Müller-Wenk and Brandão, 2010), which leads to biased results on overall LULUC emissions. In reality, physically occurring carbon fluxes will follow a logarithmic function with high initial rates of losses which are levelling off after a certain number of years after LUC, depending on climate and soils (e.g. West et al., 2004). LUC-related emissions go along with low rates of carbon losses from LU (see publication 3, section 2.3, 'Methods for estimation of emissions from

LULUC and their effects on emission loads of food and feedstuffs'). While carbon emissions from direct LUC from tropical forest to cropland in Brazil are assumed to result in 740 tons of CO_2 per hectare (37 tons annually over a period of 20 years) according to the PAS2050-standard (BSI, 2008; based on RFA (Office of the Renewable Fuels Agency), 2008) as applied in publication 2, the method by Müller-Wenk and Brandão (2010) calculates only 46.2 tons of CO_2 for total LUC-CO_2 over the whole time series. Expressed per kg DM of Brazilian SBME from former tropical forests with a 20 years accounting period, direct LUC-emissions of 0.961 and 15.397 kg CO_2-eq result following Müller-Wenk and Brandão (2010) and RFA (2008), respectively. Hence, the former report only 6 % of GHGE from LUC as compared to the latter. On the opposite, the methodological approach which was followed by Müller-Wenk and Brandão (2010) for land occupation (LU) results in GHGE of 0.616 kg CO_2-eq per kg DM of soybean, which highly exceeds the 0.119 kg CO_2-eq reported in publication 3 (section 2.3, Table 3) as an average for Brazil (20 years accounting period).

Following the method described by Müller-Wenk and Brandão (2010), emissions from LULUC for land converted from grassland in temperate climates can be estimated. This results in 410 kg CO_2 released due to LU per hectare and year, which translates into 0.103 kg CO_2 per kg barley with an estimated yield of 4 tons of DM. This is substantially higher than the approximation of 0.050 kg CO_2 per kg for barley produced in Austria or Germany (publication 1, section 2.1, Material and Methods, 'Sources of emissions'). On the contrary, the approach followed by Müller-Wenk and Brandão (2010) results in only 0.289 kg CO_2 per kg barley from LUC for newly converted land, whereas the methods from BSI (2008) and RFA (2008) would result in 1.750 kg CO_2 per kg barley produced in Germany (no values are available for Austria).

In publication 3, two default values are suggested for LUC-related accounting periods: 10 and 20 years. For the Brazilian soybeans used as feedstuff, LULUC-related GHGE decrease by about 23 % if 10 years instead of 20 years are implemented as accounting period, mainly due to reduced prevalence of soybean production on the newly deforested land. Based on the results presented in publication 3, the effect of different accounting periods may in the case of broilers result in a difference in GHGE load of the product in a magnitude of 20 %. Besides the need for accurate data on the quantity of carbon emitted from the respective

land areas, specific accounting periods should be defined which should be based on physically occurring LUC-emissions if data was available (see publication 3).
Very long accounting periods (as suggested e.g. by Kool et al., 2009; see publication 3) may still lead to LUC's effect on overall GHGE per kg product (e.g. raw milk) similar to publication 1, despite markedly different carbon footprints for specific feedstuffs. The reasons for this are generally lower LUC-emissions per kg of feed which may compensate for the higher proportion of feedstuffs which have to be accounted for; due to the longer accounting period, higher proportions of areas for production are related to LUC-emissions. However, a scenario which includes substantially longer accounting periods than used herein, would result in lower LUC-related GHGE loads for feedstuffs imported from Latin-America; yet these would be compensated by emissions which have to be included for a proportion of grains locally produced in Austria, where areas were occupied by agriculture a longer time ago (but within the accounting period suggested by Kool et al. (2009)).
On the other hand, very short accounting periods of e.g. one year would hypothetically result in physically occurring CO_2 fluxes from soil being still emitted after the end of the accounting period. Additionally, short accounting periods would lead to the phenomenon that converted areas are relieved from LUC-caused emissions after a short time period and would provide 'cleaner' products after markedly peaking LUC-related emissions, although the majority of the released CO_2 still remained in the atmosphere.

It is important to note that all LULUC-related emissions reflected in this thesis account for direct LUC only and do not consider indirect LUC (iLUC) which may occur elsewhere on the globe (especially outside national boarders) due to outcompeted and regionally shifted production of agricultural goods (see publication 3). For specific products' CFs or GWPs (LCAs), these effects of iLUC could be of tremendous importance and should be considered in future studies for products with a varying relevance at global markets. However, a suitable method for iLUC's GHGE leads to a further increasing complexity of products' CFs or GWPs. This type of LUC is not related to countries/regions with LUC anymore, but would be related to markets. Hence, market-based models and data would be required for an extensive and profound estimation of iLUC-related emissions, which are not yet covered by state-of-the-art methodology and the respective databases. Consequently, this approach was not covered in this thesis, as an adequate incorporation of iLUC would have gone far beyond the scope of this work.

3.5 Implications, conclusions and outlook

The main aim of this thesis was to adequately integrate emissions from sources that were not considered in most previous studies, and particularly from LULUC, into product carbon footprints of agricultural goods, especially for milk from typical Austrian production systems. Hence, publication 1 (section 2.1) presents the first model calculation, in which GHGE per kg raw milk were estimated, including effects from LULUC. The extension of system boundaries which is required for an up-to-date approach, resulted in advantages for organic milk production systems, which hardly utilise the LULUC-burdened soybean meal. The regional location of the farm was identified as another very important driver, exerting a great impact on emission loads per kg of raw milk, which exceeds the effect of organic or conventional production methods.

In publication 2 (section 2.2) the consequences of a substitution of feedstuffs which are specifically loaded with high GHGE from (LU)LUC are discussed. Although the mitigating effect of replacing these feed components will be substantially higher for monogastric animals, substitution of these feedstuffs is also one of the most important mitigation options for GHGE in dairy production.

In publication 3 (section 2.3), suggestions are derived for an adequate method for estimation of GHGE from LULUC. It is suggested that GHGE which are included in carbon footprints and global warming potential should be restricted to physically occurring fluxes of greenhouse gases and should exclude hypothetical sources.

Therefore, the work presented herein is assumed to meet the demands of an adequate estimation of GHGE from dairy production with special emphasis laid on direct LUC. However, it lacks an estimation of indirect LUC, which should be reflected in future studies. Furthermore, future studies should not concentrate on GHGE only, but should also cover other important aspects of sustainability: biodiversity, water consumption and the potential impairment of water quality may serve as examples. Criteria for the economic and social-ethical aspects may include income or break-even points, animal-welfare and labour conditions etc.

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