MITIGATING THE IMPACT OF AGRICULTURE ON AIR QUALITY AND CLIMATE CHANGE

SOLUTIONS FOR IMPROVED NITROGEN MANAGEMENT

IFOAM EU GROUP

FiBL

PUBLISHER:

IFOAM EU GROUP

IFOAM EU Group
Rue du Commerce 124, BE - 1000 Brussels, Belgium
Phone: +32 2280 1223 - Fax: +32 2735 7381
info@ifoam-eu.org
www.ifoam-eu.org

Authors: Beatrice Steinemann, Andreas Gattinger, Maike Krauss, Alfred Berner, Florian Leiber, Veronika Maurer, Matthias Meier, Bernadette Oehen, Lin Bautze, Urs Niggli
Pictures: Thomas Alföldi, Lukas Pfiffner, FiBL
Advisory Committee: Eric Gall and Stephen Meredith
Publication year: 2015
Drafting, editing and layout: FiBL Projekte GmbH
Printing: jcbgam printing solutions

List of abbreviations:

AA – Amino Acids
AFOLU – Agriculture, Forestry and Other Land Use
C – Carbon
CH_4 – Methane
CO – Carbon monoxide
CO_2 – Carbon dioxide
CP – Crude Protein
DM – Dry Matter
eq – equivalent
GHG – Greenhouse Gas
GWP – Global Warming Potential
LUC – Land-Use Change
LULUCF – Land Use activities and Land-Use Change and Forestry
N – Nitrogen
N_2 – Nitrogen gas
NH_3 – Ammonia
NH_4 – Ammonium
NO – Nitric oxide
N_2O – Nitrous oxide
NO_2 – Nitrite
NO_3 – Nitrate
NMVOC – Non-Methane Volatile Organic Components
O_3 – Ozone
PM – Particulate Matter
SOM – Soil Organic Matter
VFA – Volatile Fatty Acids

CONTENT

1. GHG EMISSIONS FROM THE AGRICULTURAL SECTOR

"Anthropogenic greenhouse gas emissions have increased since the pre-industrial era, driven largely by economic and population growth, and are now higher than ever. This has led to atmospheric concentrations of carbon dioxide, methane and nitrous oxide that are unprecedented in at least the last 800 000 years. Their effects, together with those of other anthropogenic drivers, have been detected throughout the climate system and are extremely likely to have been the dominant cause of the observed warming since the mid-20th century (Pachauri et al., 2014)." Global annual anthropogenic greenhouse gas (GHG) emissions rose by 10 Gt CO_2 eq (equivalent) between 2000 and 2010. Although most of these increasing emissions come from other sectors (see Figure 1), the AFOLU (Agriculture, Forestry and Other Land Use) sector contributed significantly to overall emissions (Edenhofer et al., 2014). The situation in Europe is different as agriculture is only responsible for just over 10 % of the GHG emissions as shown in Figure 2.

Greenhouse Gas Emission by Economic Sectors

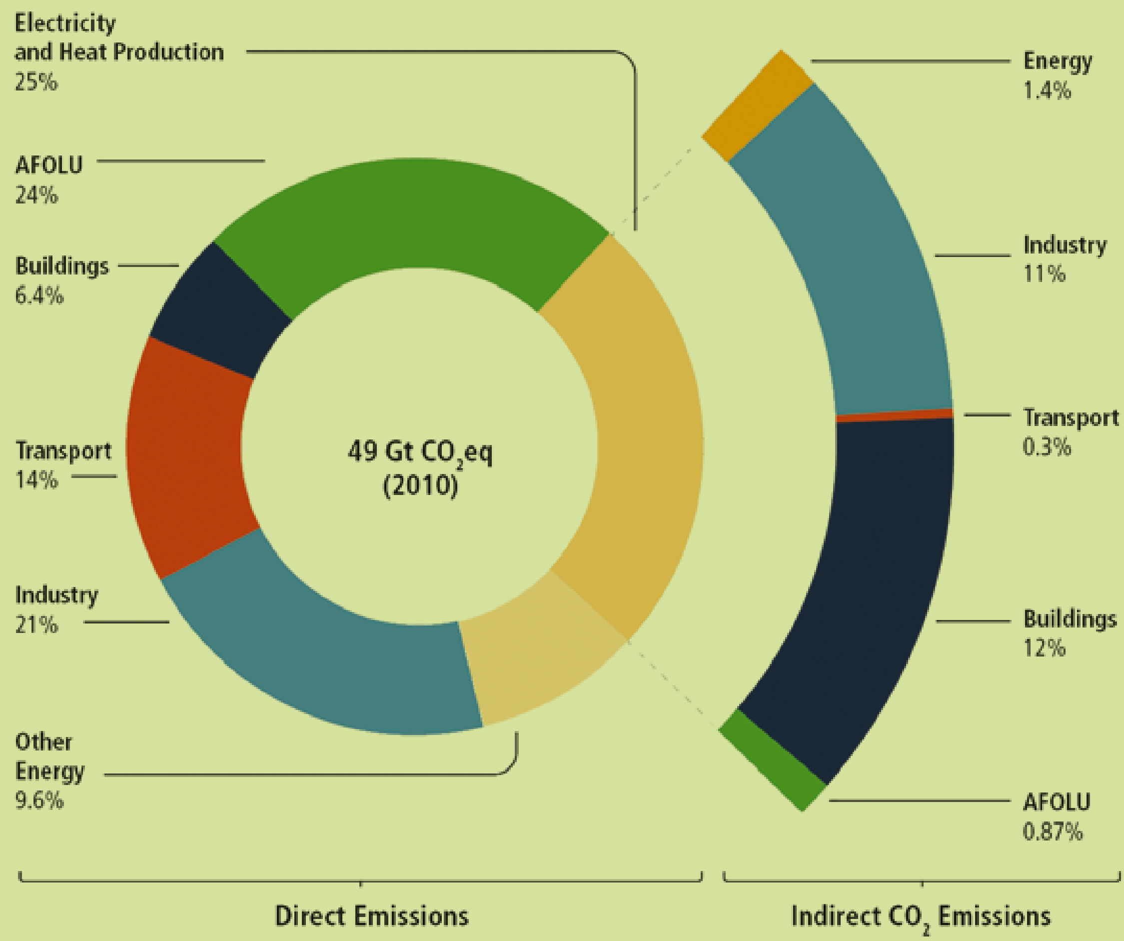

Figure 1: Total global anthropogenic greenhouse gas emissions (Gt CO_2 eq/yr) by economic sector (direct and indirect emissions (Edenhofer et al., 2014)

The global AFOLU sector accounts for about a quarter (~ 10 - 12 Gt CO_2 eq/yr) of net anthropogenic GHG emissions. The sector's emissions come mainly from deforestation, agricultural emissions from soil, rice fields, nutrient management, livestock and fossil fuel use. Organic and inorganic material provided as inputs or outputs in the management of agricultural systems are typically broken down by bacterial processes, releasing CO_2, CH_4 and N_2O to the atmosphere. The agricultural sector is therefore the largest contributor to global anthropogenic non-CO_2 GHGs, accounting for 56 % of emissions in 2005 (Smith et al., 2014). At the same time, the sector plays a key role for food security and sustainable development. It is thus essential to provide resource-efficient, air quality and climate-smart solutions within this sector in order to achieve global sustainable development in line with other political goals (Aneja et al., 2009; Erisman et al., 2008). Figure 2 shows the GHG emissions for 2012 in Europe: overall emissions (all sectors including "Land Use activities and Land-Use Change and Forestry", LULUCF), LULUCF, agricultural emissions, and within agriculture enteric fermentation, manure management and agricultural soils.

One way, to reduce overall GHG emissions from agriculture is to improve cropland management and restore organic soils. As these measures represent, according to Edenhofer et al (2014), the most cost-effective mitigation options, a combination with supply-side measures can contribute to a sustainable development of farming. On the supply side, emissions from land-use change (LUC), land management and livestock management can be reduced, terrestrial carbon stocks can be increased by C-sequestration in soils and biomass, and emissions from energy production can be saved through the substitution of fossil, non-renewable energy carriers by renewable ones. There are significant regional differences in terms of mitigation potential, costs, and applicability, due to differing local biophysical, socioeconomic and cultural circumstances. It is widely acknowledged that mitigation measures within the agricultural sector are necessary to avoid a further increase of mainly nitrogen-related emissions, while meeting the growing global demand for animal-based food. The production and emissions of N_2O are closely linked to the efficiency of nitrogen (N) utilisation within the major pathways of a livestock system – that is, animals, manure, soil

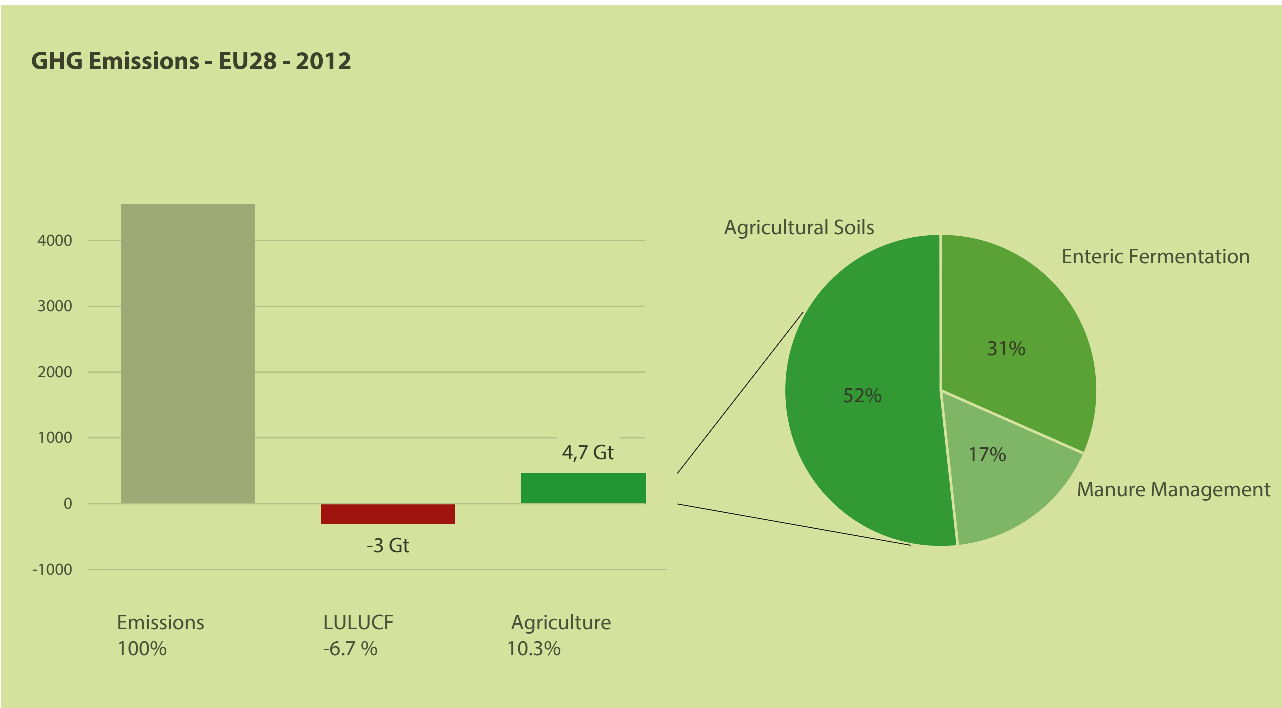

Figure 2: GHG emissions from the EU 28 countries in 2012: All sectors; Land Use, Land-Use Change and Forestry (LULUCF); agriculture, and within agriculture enteric fermentation, manure management and soils.
Source: www.eea.europa.eu/publications/european-union-greenhouse-gas-inventory-2014

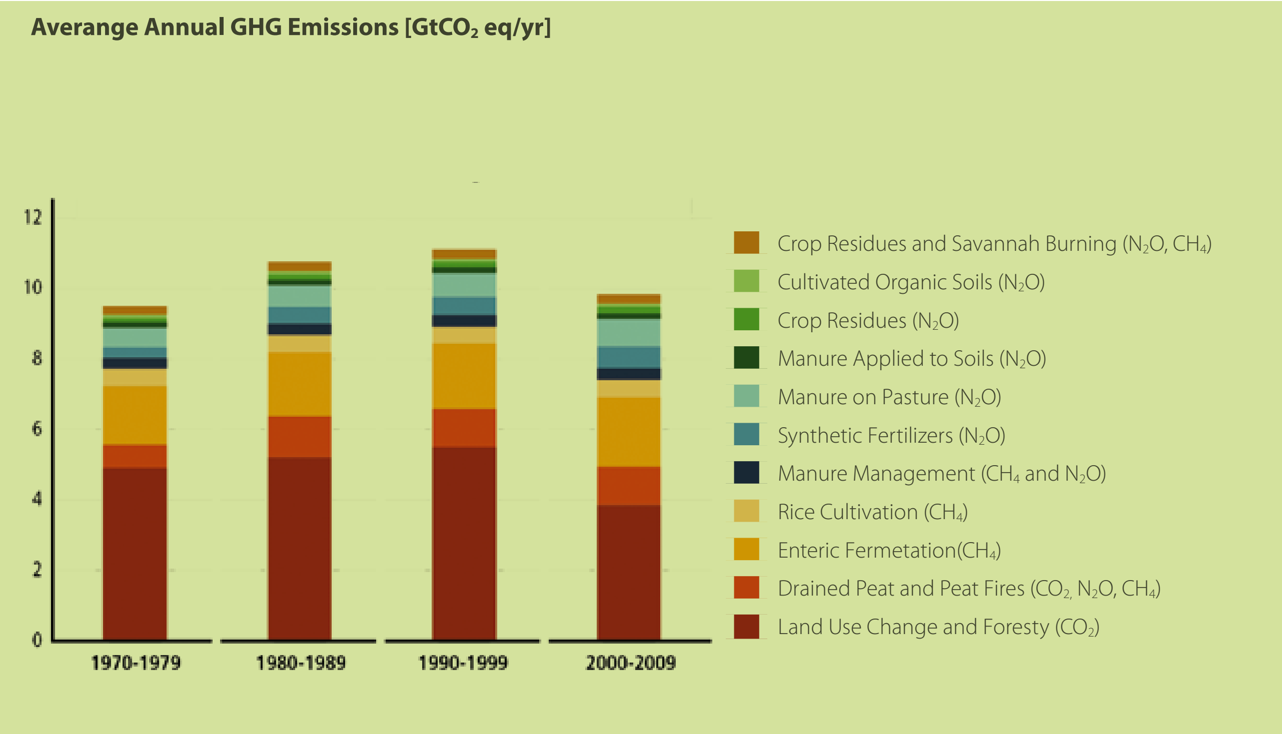

Figure 3: Global trends in total GHG emissions from AFOLU activities between 1970 and 2009 (Gt CO_2 eq/yr) (Smith et al., 2014)

and crops for feedstuffs. Besides production-driven mitigation options, demand-driven options should be considered as well. Eating less but high quality animal products is often seen as a possible solution to reduce the environmental impact of the livestock sector. This consumer choice not only depends on the environmental impact of agricultural production, but also on other sustainability issues such as animal welfare, product quality and cost price (de Vries and de Boer, 2010). Reducing food waste would also be an important mitigation option as more than 100 million tons of food is wasted annually in the EU (2014 estimate). All actors in the food chain have a role to play in preventing and reducing food waste, from producers to processors, marketers and finally consumers (http://ec.europa.eu/food/safety/food_waste/index_en.htm). Low world market prices force farmers to produce large amounts of food on an industrial scale, which highly stresses the environment. It depends on everybody how far we can reduce GHG releases from the food chain.

Anthropogenic land-use activities (e.g., management of croplands, grasslands) and changes in land use/cover (e.g., conversion of grasslands, peatlands to cropland) cause changes superimposed on natural gaseous fluxes. Agricultural activities lead to non-CO_2 emissions primarily, e.g. CH_4 from livestock or N_2O from manure storage, agricultural soils and biomass burning. Sector activities are both sources (e.g., peatland drainage) and sinks of CO_2 (e.g., management for soil carbon sequestration). Figure 3 shows global trends in total GHG emissions from AFOLU activities between 1970 and 2009 (Smith et al., 2014).

This paper focuses mainly on reactive N emissions (N_r) in form of ammonia (NH_3), nitrous oxide (N_2O) and nitrate (NO_3) as indirect and direct GHG, respectively, but will also take into account methane (CH_4) emissions caused by the agricultural sector. CH_4 plays a major role in livestock production systems as it is an unavoidable byproduct of ruminal fibre fermentation. However, it needs further to be considered that reducing one GHG can lead to an increase of another one, e.g. reducing NH_3 loss may increase nitrate NO_3 leaching and denitrification. The global warming potential (GWP) per unit mass of N_2O is about 15 times higher than that of CH_4 and 310 times higher than that of CO_2 (http://unfccc.int/ghg_data/items/3825.php).

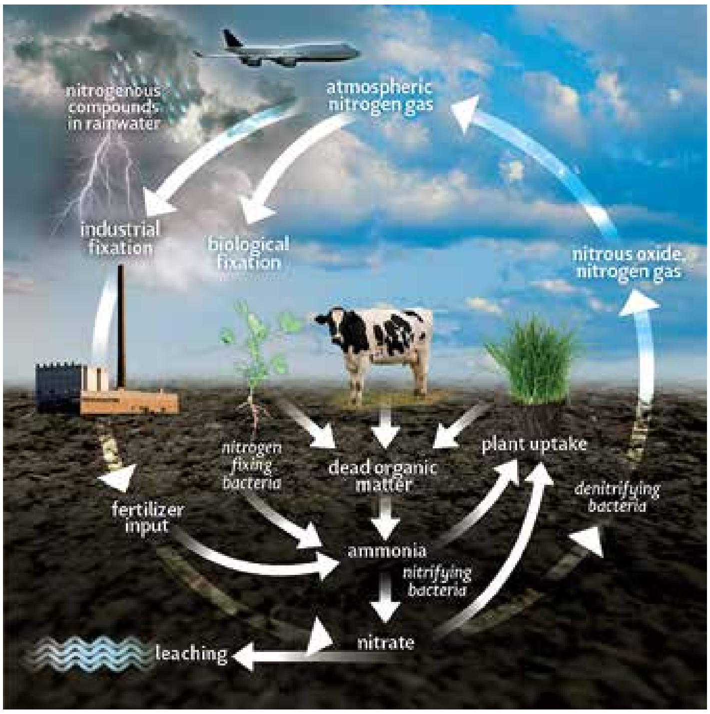

Figure 4: Nitrogen cycle; Source BBSRC.

1.1 THE NITROGEN CYCLE AND AGRICULTURE

The main agricultural emissions are N_2O, NO_3 and NO_x from agricultural soils and NH_3 from livestock farming. N emissions change the greenhouse gas balance; they also affect water quality (through eutrophication) and air quality (NH_3 and NO_x).

The nitrogen cycle and agricultural production are interlinked in several ways, as shown in Figure 4. Nitrogen has different forms (see Box), such as soluble nitrogen gas (N_2), soluble nitrate (NO_3-), nitrite (NO_2-), ammonia (NH_4+) and nitrous oxide (N_2O). These forms occur within the global biogeochemical cycles as presented in Figure 4. Nitrogen in the form of N_2O contributes as a GHG to an average increase of global temperatures; nearly 80% of N_2O releases are caused by the agricultural sector (Pachauri et al., 2008). Human technologies led to the modification of atmospheric N_2 into reactive forms (N_r) via industrial fixation of atmospheric N_2 to NH_3 (~80 Mt N yr-1), agricultural fixation of atmospheric N_2 via cultivation of leguminous crops (~40 Mt N yr-1), fossil-fuel combustion (~20 Mt N yr-1) and biomass burning (~10 Mt N yr-1). The primary purpose of N conversion is to increase food production via fertiliser use. The negative side effects include global warming, soil and the pollution of waterways, potable water and coastal zones due to the leaching of nutrients. Thus the agricultural sector alters natural biogeochemical cycles via anthropogenic inputs or unsustainable farming practices. This jeopardises the ecosystem services provided by terrestrial, aquatic and marine systems and pushes the planetary biogeochemical flow boundaries to critical continental or even global thresholds (Rockström et al., 2009).

The primary effects of N_r inputs are increased emissions of N trace gases (N_2O, NH_3) to the atmosphere. The processes driving the biosphere-atmosphere exchange of these compounds, such as nitrification and denitrification (N_2O and NO) or volatilisation (NH_3) depend significantly on the availability of N_r in the plant-soil system. Thus, increased N_r inputs to agricultural systems (with livestock farming systems having the highest N_r use intensity in Europe) lead to increased losses of N trace gases (NH_3, N_2O) at the site of Nr input (Butterbach-Bahl et al., 2011).

Agricultural perturbations to the global nitrogen cycle, directly and indirectly, lead to enhanced biogenic production of nitrous oxide (N_2O). Direct pathways include microbial nitrification and denitrification of fertiliser, plant residues and manure nitrogen that remains in agricultural soils or animal waste management systems. Indirect pathways involve nitrogen that is removed from agricultural soils and animal waste management systems via volatilisation, leaching, runoff or harvest crop biomass (Nevison, 1998). Further indirect impacts are fossil fuel use in farm operations, the production of agrochemicals and the conversion of land for agricultural production.

1.2 GASEOUS EMISSIONS – THEIR IMPACT ON THE ENVIRONMENT AND AIR QUALITY

Air quality, ecosystem exposure to nitrogen deposition, and climate change are intimately coupled problems. At present, particulate matter (PM) and ground-level ozone (O_3) are Europe's most problematic pollutants in terms of harm to human health, followed by benzo(a)pyrene (BaP) and nitrogen dioxide (NO_2) (EEA, 2014). Primary PM mainly originates from the burning of agricultural waste. This practice is banned in cross-compliance rules under the Common-Agricultural Policy (CAP) and has additionally been prohibited in several European States. In terms of damage to ecosys-tems, the most harmful air pollutants are O_3, NH_3 and NO_X (EEA, 2014). Emissions of reactive trace gases, generated in the burning of fossil fuels and biofuels and volatilised from agricultural processes, cause a number of environmental problems. Ozone (O_3) forms from the photochemical oxidation of methane (CH_4), carbon monoxide (CO) and non-methane volatile organic components (NMVOC) in the presence of nitrogen oxides ($NO_x = NO + NO_2$) (Dentener et al., 2006). CH_4 is also the most abundant reactive trace gas in the troposphere; its reactivity is important to both tropospheric and stratospheric chemistry (Wuebbles and Hayhoe, 2002). O_3 in the troposphere is an important greenhouse gas and is also toxic to humans, animals and plants (Dentener et al., 2006). Emission of reactive N species to air and waters may affect life on Earth in various ways (Sutton et al., 2011). Observed effects include pollution of groundwater due to NO_3 leaching, eutrophication of surface water, terrestrial eutrophication leading to a decrease in species diversity, soil acidification, global warming, impacts on human health and plants, and stratospheric ozone destruction due to N_2O (Erisman et al., 2007; Velthof et al., 2014). NH_3 emissions may not change the greenhouse gas balance directly, but do serve as a substrate for the microbial formation of N_2O through nitrification/denitrification processes. Moreover, as a constituent of atmospheric deposition NH_3 contributes to eutrophication of various natural ecosystems (Webb et al., 2014), and causes odour problems around livestock production systems that are often a source of conflict with local residents. NH_3 emissions to air are a major threat to human health, causing cardiovascular and respiratory diseases (www.unece.org/index.php?id=37612).

NH_3 mainly comes from manure produced by livestock and from mineral nitrogen fertilisers. On 11 December 2014, Parties to the UNECE (United Nations Economic Commission for Europe) Convention on Long-Range Transboundary Air Pollution adopted a new Ammonia Framework Code to help countries reduce NH_3 emissions from agriculture. European emissions of NH_3 dropped by 25 % between 1990 and 2011. Agriculture was responsible for 94 % of NH_3 emissions in 2011 (www.eea.europa.eu/data-and-maps/indicators). The reduction is primarily due to a reduction in livestock numbers, changes in the handling and management of manures and from the reduced use of nitrogenous fertilisers. The main source of NH_3 is the rapid hydrolysis of urea in urine by urease, leading to ammonium (NH_4). Another source of NH_3 is the degradation of undigested proteins. This pathway is rather slow. Urease activity is driven by temperature, with low activity below 5-10°C and above 60°C. Urease activity is further affected by pH with an optimum between 6 and 9, while animal manure pH ranges between 7.0 and 8.4 (Philippe and Nicks 2013). Odour emission increases not only with indoor temperature but also with barn ventilation rate and animal activity (Philippe and Nicks, 2013; Schauberger et al., 2013). The formation of N_2O occurs during incomplete nitrification/denitrification processes that normally convert NH_3 into non-polluting N_2 (Philippe and Nicks, 2013).

DIFFERENT FORMS OF NITROGEN

Nitrogen gas (N_2) makes up 78 per cent of the atmosphere. It is not directly available for use by plants but is directly used in biological nitrogen fixation by free-living or symbiotic diazotrophic bacteria and industrial fertiliser manufacture.

Ammonia (NH_3) is an odorous gas that is unavailable to plants but is a major threat to human health. It is produced when microorganisms break down organic nitrogen products, such as urea and proteins in manure. This decomposition occurs in both aerobic and anaerobic environments.

Nitrate (NO_3–) is an ionic form of nitrogen and is the most common form available to plants. In this form, nitrogen is mobile, leachable and usually the end product of mineralisation.

Ammonium (NH_4+) is also an ionic form of nitrogen available to plants. Plants use less energy for uptake in this form compared to NO_3. Nitrogen in this form is less likely to be lost from the soil than other forms.

Nitrite (NO_2–) is an intermediate in the conversion of NH_4 to NO_3. Nitrogen in this form is not available to plants and is more prone to be lost from the soil than either NO_3 or NH_4.

Nitrous oxide (N_2O) is like CH_4 in that it is a long-lived greenhouse gas responsible for increased radiative forcing to the climate system. N_2O has an atmospheric lifetime of about 114 years, and each molecule of N_2O has a direct global warming potential around 300 times that of CO_2 (Andersen et al., 2014). Soil nitrogen can be lost in this form through denitrification (Erisman et al., 2007).

Nitric oxide (NO) is also a gas, and nitrogen in this form is lost through denitrification. It may be harmful to the ozone layer.

1.3 IMPACTS OF DIFFERENT LIVESTOCK SYSTEMS ON THE ENVIRONMENT

Livestock management and associated food production is linked with GHG emissions and further impacts on the global and regional nitrogen cycle. Global consumption of animal-derived products (meat, milk, eggs) has risen in recent years and is likely to increase further in the future. This process is driven on the one hand by global population growth and on the other hand by the opening of new markets in the developing world (Fiala, 2008). Due to low world market prices, industrial farms have emerged. Organic and low-input farms, in contrast, aim to close nutrient cycles at farm level (Gattinger et al., 2013).

Industrial food production and animal products entail high environmental costs because of large emissions of greenhouse gases from soil, plant, and livestock systems (see Figure 3), as well as substantial energy consumption in growing (fertiliser production and usage for animal fodder), transporting (fuel use), processing and refrigerating the food products (electricity use). The energy requirements and greenhouse gas emissions in production of animal-derived foods are much greater than in production of other food products such as grains, vegetables, and fruits (Stavi and Lal, 2013). In addition to direct emissions from livestock, animal-derived food production is implicated in many harmful processes such as land-use change, loss of biodiversity, excess water use, nutrient excretion, fossil energy use, competition for food and emission of greenhouse gases. At the same time, with proper management, animal-derived products offer numerous benefits such as producing food from human-inedible sources, preserving ecosystem services, promoting perennials on croplands, recycling plant nutrients and providing social benefits. Thus, livestock can be both stressors and benefactors to land and humans. The aim should be to shift the net effect from stress to benefit (Janzen, 2011). Within livestock and animal-derived food production, stresses on the environment differ according to animal physiology (fodder utilisation, reproduction rate) and the needs of the animal (fodder, space), production system (conventional, organic, and indoor/outdoor) and scale of production (intensive and extensive). As shown in Figure 5, these differences are responsible for diverse nitrogen footprints within the agricultural sector. Herbivores such as the common agricultural animals (ruminants and non-ruminants) are a significant source of N_2O emissions. Herbivores alone account for a large share of manure-related N_2O emissions, as well as soil-related N_2O emissions through the use of grazing land and land for feed and forage production. The N_2O emission of the total livestock sector in the EU-27 rose in the past few decades to 120 Mt CO_2 eq, of which 75 % was emitted from herbivore production systems. Within the herbivores, the dairy and beef sectors were responsible for 90 % of total N_2O emissions (Schils et al., 2013). This comparison needs to take into account that ruminants feed mainly by roughage, maintain grasslands and contribute to maintaining sequestered carbon in soils.

2. EMISSIONS FROM THE LIVESTOCK SECTOR

The livestock sector contributes to European GHG emissions as shown in Figure 2. The beef and dairy subsectors contribute 60 % of all livestock production emissions as shown in Figure 5. The total GHG mitigation potential from the livestock sector in Europe is 101 - 377 Mt CO_2 eq. Between 12 and 26 Mt CO_2 eq can be mitigated if previous cropland is converted to a permanent vegetation type, and between 39 and 79 Mt CO_2 eq if the production and decomposition of wasted animal food is avoided (Bellarby et al., 2013). A life-cycle reduction in GHG emissions based on a life cycle approach from livestock can be based on technical (e.g. manure application techniques), management-based (e.g. changes in grazing/housing patterns) and system-oriented approaches (e.g. shift from conventional to low input or organic farming) (Novak and Fiorelli, 2010), or can be achieved by reducing demand for animal products. Reduced consumption of meat products would also be beneficial for public health through decreased cardiovascular disease connected with intake of saturated fat, and through decreased risk of colorectal cancer connected with the intake of red meat (Cederberg et al. 2013). Policies will need to take regional differences into account as well as the needs of both producers and consumers, while operating across sectors such as public health and consumption behaviour, agricultural production and marketing and trade (Bellarby et al., 2013). Projections by FAO indicate that animal numbers will increase by 40 % between 2000 and 2030. These trends suggest a strong increase in total N_2O emission from animal production systems in the near future, which is opposite to the objec-tives of the Kyoto Protocol (Oenema et al., 2005). Losses of NH_3 occur during slurry application, housing, slurry storage, grazing, fertiliser application and from crops, in descending order of importance (Bussink and Oenema, 1998).

2.1 RUMINANTS

The largest livestock-related GHG emissions in the EU come from the dairy sector followed by beef. Together, these sectors account for more than 70 % of GHG emissions from livestock production (Lesschen et al., 2011); see Figure 5. Cattle dominate ruminant husbandry in Europe and constitute the most important source of anthropogenic CH_4. In addition to enteric (animal-derived) CH_4, excreta are another source of CH_4, especially when stored anaerobically. CH_4 originates from the anaerobic degradation of organic matter performed by mesophilic/thermophile bacteria with an optimal pH close to neutrality (Philippe and Nicks, 2013) and is an unavoidable by-product of ruminal fibre fermentation.

2.1.1 PRODUCTION SYSTEMS

Organic farming practices are considered to have a significant potential for GHG mitigation through enhancing soil carbon stocks and reducing soil-derived N_2O emissions. They also provide various co-benefits including capacities for climate change adaptation. On the other hand, there is a risk that in organic systems higher sequestration rates and lower GHG emissions may be compensated by less favourable CO_2 eq balances resulting from lower yields in organic systems. Then again, some studies show that the GHG-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 and has greater overall system efficiency. GHG emissions of conventional dairy farming are high due to the effects of land-use change and transportation of feedstuffs (Hörtenhuber et al., 2010). While organic farming as such can be a promising GHG mitigation measure, recent studies have found a high variability between farms (organic and conventional), even if they are of the same farm type or in the same region (Kristensen et al., 2011). This implies a high potential for optimising farm management with respect to GHG mitigation potential on organic and conventional farms alike (Schader et al., 2014). A number of studies have been performed to estimate carbon footprints in organic and conventional production systems. No major systematic difference between the different production systems seems to be present (Cederberg, 2004; Kristensen et al., 2011; Thomassen et al., 2008). The impacts per area farmed land are usually less in organic systems, but per quantity produced impacts are often higher. Comparative life-cycle assessments of agricultural products from organic and conventional farming systems often do not adequately differentiate the specific characteristics of the respective farming system. The most critical points are the N fluxes influencing acidification, eutrophication, global warming potential, and biodiversity (Meier et al., 2015). Pasture-based systems can be considered not only animal friendly but also favourable in terms of climate impacts, as they emit less GHG than any other cattle keeping system (O'Mara, 2012; Taube et al., 2014). Pasture can be expected to protect nitrogen very well against emission

processes. Because of the permanent nature of grassland, no soil cultivation is necessary and therefore tillage-associated GHG emissions are saved. The use of fossil energy can also be reduced if feeding is generally based on the utilisation of pastures and the avoidance of feedstuffs transported over long distances. In regions which are dominated by grassland, cattle and other ruminants are an essential element of regional agricultural food production – here system changes would have tremendous socio-economic and high ecological costs (Hörtenhuber et al., 2010; Zehetmeier et al., 2012). Options that increase lifetime performance or reduce the replacement rate of cattle are likely to reduce GHG emissions at the farm level. This could, for instance, be achieved by dairy cows resistant to illness and better adapted to the environment of their farm rather than high-yielding animals (Hermansen and Kristensen, 2011; Novak and Fiorelli, 2010). This means that the decision to replace cows should not be based solely on economic considerations or fertility. Animal breeding may reduce N_2O emissions through the improved conversion of feed N into animal N, i.e. high feed protein conversion efficiency. A high conversion efficiency may, however, also be achieved via extensive (grass-based) feeding systems (Leiber et al., 2015, in publication). Enhanced health and fertility will contribute to production efficiency, especially in extensive pastoral systems (Schils et al., 2013). Within dairy production the major product is milk, but an important co-product is beef. Keeping dual purpose cows can reduce GHG emissions, as fewer animals are needed for rearing beef and dairy cattle.

2.1.2 HOUSING

Emissions from livestock housing are a major source of pollution within agriculture and usually the second largest source of NH_3 after slurry application. Housing systems for cattle vary across Europe. While loose housing is most common and animal friendly, dairy cattle are still kept in tied stalls in quite a few regions. In loose housing systems all or part of the excreta is collected in the form of slurry. In systems where solid manure is produced (such as straw-based systems), it may be removed from the house daily or remain there for up to the whole season, such as in deep litter stables. The system most commonly researched is the "cubicle house" for dairy cows, where NH_3 emissions arise from soiled slatted and/or solid floors and from manure in pits and channels beneath the slats/floor. Cattle held in tied stalls emit less NH_3 than in loose housing systems, because a smaller floor area is soiled

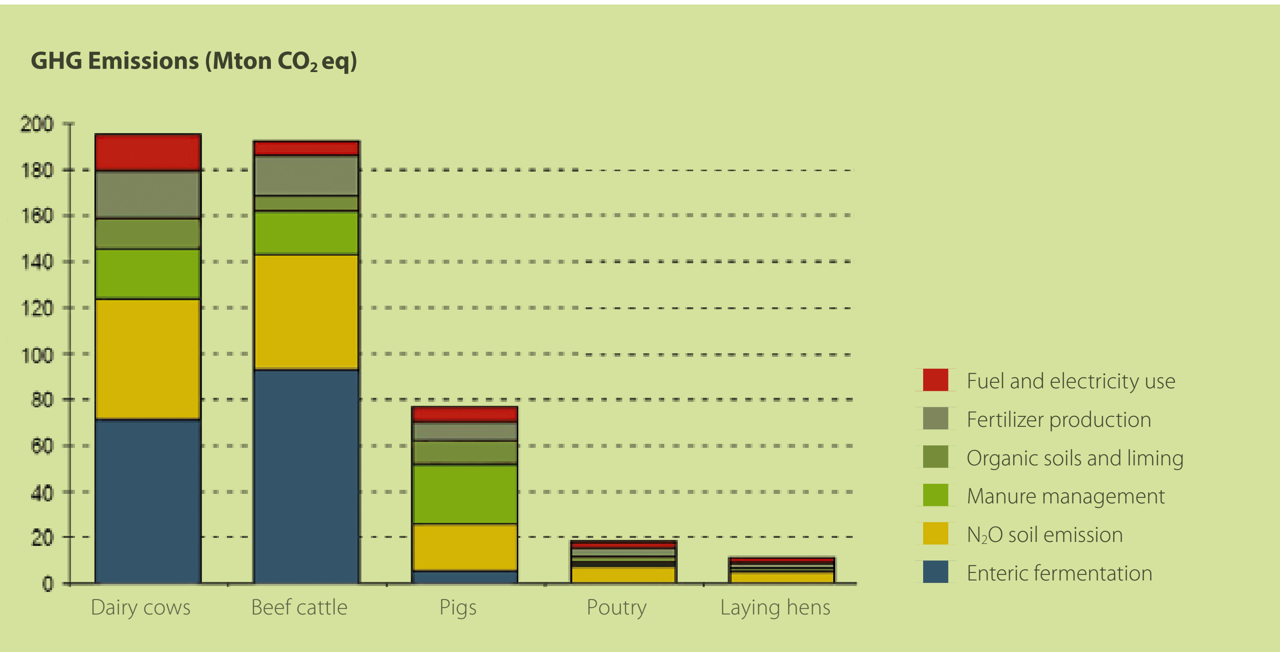

Figure 5: Total greenhouse gas emissions from the various emission sources associated with livestock production in the EU-27 (Lesschen et al., 2011)

with dung and urine. However, for animal welfare reasons tied systems are not recommended unless daily exercise periods are ensured. The potential to increase grazing is often limited by soil type, topography, farm size and structure (distances), climatic conditions, etc. It should be noted that grazing of animals may increase other forms of N emissions (NO_3-N-leaching, N_2O emissions) (United Nations, 2014). In northern European conditions, grazing is generally not possible during late autumn and winter and cows are kept in buildings during this period. Slurry is therefore collected from cowsheds under cold conditions. Several studies have shown that CH_4 and NH_3 emission rates for slurry increase significantly with storage temperatures (Novak and Fiorelli, 2010). Animal welfare considerations tend to lead to an increase of walking area per animal, increased ventilation to lower the in-house temperature and therefore an overall increase in emissions. Changes in building design to meet the new animal welfare regulations in some countries (e.g., conversion from tied stall to cubicle housing) will therefore increase NH_3 emissions unless abatement measures are introduced at the same time to prevent this increase. Changes in existing buildings or new construction to meet animal welfare requirements present an important opportunity to introduce NH_3 mitigation measures at the same time (United Nations, 2014). Measures to reduce NH_3 emissions from buildings tend to be expensive and have the further disadvantage that, unless subsequent abatement measures are introduced, at least some of the NH_3 captured in the building may be lost during manure storage or after spreading to land (Gilhespy et al., 2009).

The floor system and, related to this, the removal of the slurry from the house are main factors in NH_3 emission rates (Novak and Fiorelli, 2010). Emission rates depend greatly on floor type and manure handling method (Jongebreur and Monteny, 2001; Zhang et al., 2005). Urea concentration in the urine, urease activity, pH, temperature, air velocity, and area of emitting surfaces (floor, pit) are parameters influencing NH_3 emission (Jongebreur and Monteny, 2001). Solid floors with a smooth surface, scraper and drain may reduce NH_3 emission from free-stall dairy cattle buildings. For buildings with slatted floors, channel scrapers are potential alternative methods for keeping ammonia emissions low (Zhang et al., 2005). The "grooved floor" system for dairy and beef cattle housing, employing "toothed" scrapers running over a grooved floor, is a reliable technique to abate NH_3 emissions. Grooves should be equipped with perforations to allow drainage of urine. This results in a clean, low-emission floor surface with good traction for cattle to prevent slipping. NH_3 emission reduction ranges from 25 % to 46 % relative to the reference system. In houses with traditional slats (either non-sloping, 1 % sloping or grooved), optimal barn climatisation with roof insulation (RI) and / or automatically controlled natural ventilation (ACNV) can achieve a moderate emission reduction (20 %) as well as increased animal welfare due to the decreased temperature (especially in summer) and reduced air velocities.

Deep litter may result in significant emissions of N_2O and CH_4, depending on the rate of litter addition and mixing. Options to reduce CH_4 emissions consist here of avoiding anaerobic conditions in the bedding (Novak and Fiorelli, 2010). Cattle have the tendency to lie on straw beds and compact the accumulated straw and faeces/urine; this creates anaerobic conditions, which inhibit nitrification and the subsequent denitrification of oxidised N. According to (Gilhespy et al., 2009; Novak and Fiorelli, 2010) the use of extra straw can reduce NH_3 emissions by reducing airflow across surfaces soiled by urine, and by immobilisation of ammonium–N (NH_4–N) by bacteria using a high C:N material as a substrate. These systems have the advantage of no subsequent increase in NH_3 losses during storage or spreading, as all the ammoniacal nitrogen is immobilised in the straw. According to Gilhespy et al. (2009), increasing overall straw use reduces NH_3 emissions by 50 %, using 33 % extra straw (4.7 kg instead of 3.5 kg per cow). No further significant benefit could be gained by increasing straw use in excess of 33 %. However, in addition to NH_3 losses from housing, a whole-system approach should be adopted to consider other N loss pathways and their effects on downstream losses. Straw-based systems producing solid manure for cattle are not likely to emit less NH_3 in animal housing than slurry-based systems. Further, N_2O and N_2 losses due to (de)nitrification tend to be larger in litter-based systems than slurry-based systems (Powell et al. 2008). The physical separation of faeces (which contains urease) and urine in the housing system reduces hydrolysis of urea, resulting in reduced emissions from both housing and manure spreading (Burton, 2007).

While increased grazing is a reliable emission reduction measure for cattle, the amount of emission reduction depends on the daily grazing time and the cleanliness of the house and holding area. In some cases, if too many animals are too long in the same meadow, grazing can contribute to increased leaching or increased pathogen and nutrient loading of surface water (Hubbard, 2004).

2.1.3 FEEDING

Optimised livestock feeding strategies decrease NH_3 emissions from manure in both housing and storage, and following application to land. Livestock feeding strategies are more difficult to apply to grazing animals, but emissions from pastures are low (United-Nations, 2014). Livestock feeding strategies can influence the pH of dung and urine. The pH of dung can be lowered by increasing fermentation in the large intestine. This increases the volatile fatty acids (VFA) content of the dung and causes a lower pH. The basic function of milk-producing ruminants is to convert low-quality non-competitive feed sources into high-quality protein for human consumption. The amount and quality of protein absorbed from the small intestine can limit milk production. However, this fact leads to high feeding recommendations for crude protein (CP) in ruminants and in consequence to poor N utilisation on the rumen level. Oversupply with dietary protein leads to increased NH_3 production in the rumen and consequently higher N emissions via the urine. A slight undersupply with dietary protein may counterbalance this problem and thus increase protein efficiency (Leiber et al., 2015) and significantly limit urinary N excretion (Nousiainen et al., 2004; Spek et al., 2013). Of all single dietary and animal factors evaluated to predict N excretion in urine, milk urea and dietary CP concentration were by far the best predictors according to (Nousiainen et al., 2004; Spek et al., 2013). Thus, adjustments towards lower recommendations for dietary CP supply and milk urea values appear to be justified (Furger et al., 2013; Leiber, 2014) and could limit volatile N-molecules in cattle urine.

Phase feeding is an effective and economically attractive measure even if it requires additional effort. Young, growing animals and high-productive animals require more protein concentration than older, less-productive animals (United-Nations, 2014). More protein is also needed at the beginning of lactation than for the last 80 to 100 days. Phase feeding can be applied in such a way that the CP content of dairy diets is gradually decreased from 16 % of dry matter (DM) just before parturition and in early lactation to below 14 % in late lactation and during the main part of the dry period. Phase feeding can also be applied in beef cattle in such a way that the CP content of diets is gradually decreased from 16 % to 12 % over time (United-Nations, 2014).

Low-protein animal feeding is one of the most cost-effective and strategic ways of reducing NH_3 emissions. Controlling the N content of the diet is, however, not easy in dairy farming systems based on herbage, because the proportion of legumes can vary greatly between swards through the seasons and between years. The N content of legumes and grasses can be higher than that required for optimum animal nutrition (Novak and Fiorelli, 2010). Thus, suitable feeding management and nutrient balancing is a key issue in forage-based feeding systems as well. This is, however, also achievable with low concentrate systems if a sufficient diversity of different roughages is produced and provided. There seem to be no animal health and animal welfare implications if CP in dairy cattle diets are moderately decreased (Furger et al., 2013). Low-protein animal feeding is most applicable to housed animals and more ambitious for grassland-based systems (United-Nations, 2014). Safety margins in the protein content of the diet are used to account for suboptimal amino acid ratios and variations in requirements between different animals (United-Nations, 2014).

According to (Røjen et al., 2008) grazing poses several management challenges to intensive dairy farming, namely: N utilisation, diet optimisation and seasonal variation in production level. Dairy cows and other ruminants have a unique ability to recycle urea-N in the reticulorumen, where it can be utilised in microbial protein synthesis. Under practical farming conditions dairy operations utilise only approximately 25 % of dietary N for milk, weight gain and foetal growth. Numerous diet-related factors can influence N efficiency, such as overfeeding with rumen degradable protein, unbalanced composition of amino acids and undersupply of nutrients. Mitigation technologies involving diet-based intervention are primarily aimed at reducing the amount of N excreted in urine. In intensive, highly fertilised grassland systems, farmers tend to let the cows graze in relatively young lush grass, with high protein contents. Grazing at a later stage is an option to reduce N intake and excretion. However, grazing in older grass might reduce digestibility, and thus increase CH_4 emissions. Thus, a proper grazing management, which is based on feed analyses and involves supplements with hay or energy-rich silages or concentrates to counterbalance situations of protein oversupply, is a key factor of low-emission livestock farming.

Increasing the proportion of concentrate feeds is often considered to be the primary CH4 mitigation option. However, it is unclear whether this is still valid when diets to be compared are energy balanced. In addition, non-structural carbohydrates and side effects on nitrogen emissions may be important (Klevenhusen et al., 2011). A further aspect is that replacing roughage by concentrates contradicts European environmental policy to promote extensive use of maintained grasslands ("Greening policy"), which store significant amounts of carbon in soil (O'Mara, 2012). And leave arable land for direct human consumption. Furthermore, the production and transport of concentrates generate GHG (Novak and Fiorelli, 2010). An increase in CH_4 emission from slurry was observed when cows were fed two different mixed forage-concentrate diets instead of forage-only (ryegrass/hay) diets or diets containing more than 900 g silage/kg feed. CH_4 emissions from the slurry were found to be largely increased when enteric CH_4 formation had been suppressed by lipid supplementation. This resulted in an almost complete compensation of the CH_4 mitigation achieved in the animal (Klevenhusen et al., 2011). Dietary factors such as nutrient type and degradability influencing CH_4 may also affect N emissions from manure. The forage-only diet was superior to the two mixed forage-concentrate diets, especially regarding fibre digestibility. The reasons for this were that the forage part of the mixed diets consisted of straw, which is in general characterised by high lignification and therefore lower digestibility (Klevenhusen et al., 2011). Consistent with the lack of clear diet effects on enteric CH_4 formation, there was no such effect on manure-derived CH_4. It has to be considered, however, that the diets were balanced in their contents of energy and N, which might have been the reason for the lack of an effect. Concerning N emissions from slurry, the importance of ruminal degradability of starch and protein was demonstrated (Klevenhusen et al. 2011). Forage plant species also affect CH_4 production in ruminants. Measurements of CH_4 production from grazing beef cows indicated a 25 % reduction in CH_4 losses with alfalfa-grass pastures compared to grass-only pastures. For instance, condensed tannin-containing legumes, such as sulla (Hedysarum coronarium), were shown to reduce the CH_4 emissions of dairy cows and other ruminants (Novak and Fiorelli, 2010).

Increased forage digestibility is expected to increase animal production and decrease CH_4. It appears that C4 grasses produce greater amount of CH_4 than C3 grasses and that introduction of legumes in warm climates may offer a mitigation opportunity. Legume silages may also have an advantage over grass silage due to their lower fibre content and the additional benefit of replacing organic N fertiliser. With all silages, effective preservation will improve silage quality and reduce GHG emissions. Forages with higher sugar content may reduce urinary N losses although more research is needed to support this concept. Overall, increasing forage digestibility and digestible forage intake typically decreases CH_4. Improving forage quality and optimising rumen function for higher microbial protein synthesis through feeding of a balanced diet matching the physiological stage of the animal are the most efficient ways of decreasing CH_4 emissions per unit of animal product (Hristov et al., 2013).

2.1.4 BEEF PRODUCTION

The greatest variation was observed among the global warming potential (GWP) values for production of beef (de Vries and de Boer, 2010). A reason for this is that beef is produced in a heterogeneous range of production systems. The heterogeneity of the European beef sector is reflected in terms of specialisation, types of animals (e.g. suckler cows, or bulls), and production systems. This heterogeneity depends on natural environment (soils, climate, topology), agricultural traditions, and public policies (Hocquette and Chatellier, 2011). Apart from the beef originating from culled dairy cows, there are two main categories: suckler systems, where suckler calves are reared with their mother for an extensive period (beef cattle systems) followed by a fattening period, and beef produced from bull calves primarily from dairy herds and reared in specialised fattening units (Hocquette and Chatellier, 2011; Zehetmeier et al., 2012). Beef cattle systems are traditionally based on pasture in less productive areas and relatively low feeding intensity compared to the more intensive feeding of bull calves in dairy production. This, and the fact that the feed requirement of the mother cows has to be accounted for, results in a greater dry matter intake per kilogram of beef produced in such systems (de Vries and de Boer, 2010). It is noteworthy that feed input for suckler cow production is mainly derived from non-human-edible sources – forage and hay – (Wilkinson, 2011). Furthermore, suckler cows can be farmed on less valuable land with a high percentage of permanent grassland; this is associated with other ecosystem services such as conservation of biodiversity, water quality and aesthetic value (Zehetmeier et al., 2012).

Suckler cow-calf beef systems have a low feed and energy conversion efficiency. The GHG balance is considered to be similar or more favourable in forage-based dairy cattle systems using dual-purpose breeds than in concentrate-based dairy cattle systems (Zeitz et al., 2012). However, concerning CH_4 emissions, it is important to note that maintaining milk production with fewer dairy cows but higher milk yield is mostly associated with an increase in suckler beef production, which overall leads to a less favourable GHG balance of total milk and meat production. According to (Meier et al.,

2.1.5 RECOMMENDATIONS FOR GOOD MANAGEMENT PRACTICES

It is important to make sure that all beneficial interactions in the livestock system are optimised instead of focusing only on animal productivity:

- Reducing NH_3 loss requires a whole-farm system approach, because such an approach reveals how intervening in one part may affect NH_3 losses in other parts of the system. Reducing NH_3 loss can increase NO_3 leaching and denitrification.

- Across production systems farming strategies based on either low stocking rate or with focus on high efficiency in the herd can be equally successful.

- Low-protein animal feeding, adapted to real needs, is a cost-effective, strategic way of reducing NH_3 emissions. The CP content of diets for dairy cattle should not exceed 15-16 % in DM. For beef cattle over 6 months this could be even further reduced to 12 %.

- A diet based on high forage content is recommended due to the nutrient competition of concentrate food with humans' nutrition.

- Phase feeding is an effective and economically attractive measure.

- Condensed tannin-containing legumes have been shown to reduce the CH_4 emissions of dairy cows and other ruminants.

- Optimised feed conversion at system level and use of feeds from cropping systems that increase soil carbon sequestration versus carbon emission.

- Appropriate (climate, soil, time) grazing management is a reliable emission reduction measure; the amount of reduction depends on the daily grazing time and the cleanliness of the house and holding area.

- Use of grass legume mixtures to optimise symbiotically fixed nitrogen, and reduced use of inorganic nitrogen fertilisers in pasture-based systems.

- Improving the nutrient density of forage to reduce GHG emissions from enteric fermentation.

- Beef production from calves out of milk production raised on an organic grazing system using dual-purpose cattle is more efficient than beef from suckler systems.

- Improving health, welfare, performance, fertility and therefore lifetime performance reduces the number of animals required and the associated GHG emissions.

- Breeding for improved N conversion, good health and high animal productivity.

- Avoiding anaerobic conditions in the bedding of deep litter systems. The use of extra straw can reduce NH_3 emissions by reducing airflow across surfaces soiled by urine, and by immobilising ammonium–N.

- Solid floors with a smooth or grooved surface, scraper, channel scraper and drain reduce NH_3 emissions from free-stall dairy cattle buildings.

- Optimal barn climatisation with roof insulation (RI) and/ or automatically controlled natural ventilation (ACNV) can achieve a moderate emission reduction.

2.1.6 FURTHER RESEARCH NEEDS

Simulation modelling, combining biophysical and decision models, would be useful to assess the balance of different sets of mitigation options by taking into consideration trade-offs, interaction and feed-back among practices at the farm level, and by evaluating their impact upon environmental and agronomic aspects. With such dynamic simulations, all farm management options could become optimised and farmers can apply the best individual sustainability practice. The development of nutrient balancing techniques in feeding systems with reduced concentrate use is a key task for the provision of appropriate feeding recommendations for low-input livestock agriculture.

Further practical research is needed for low-input grazing systems based on grass-clover swards with highest energy-use efficiency, taking also into account the carbon sequestration potential and CH_4 mitigation potential of tannin-rich forage plants which have a positive effect on both animal health and productivity.

Research should be targeted at practical issues, for instance the potential of precision livestock farming for mitigation (e.g.: individually adapted feeding regimes; best match between genetics and low-input environment; environmentally best animal excreta handling), or the specific efficiency of production based on roughage and fibre utilisation. There is thus a need to gather more data on the competitiveness of different low-input beef supply chains.

The impacts of global change on livestock systems should be taken into account more rigorously for all research activities in livestock production. Vigorous animals (in terms of robustness, resistance and recovery), high quality feed, and improved nutrient utilisation with more autonomous and low-external input farming systems would ensure better incomes for farmers while protecting the environment and producing typical, specific products of high quality.

2014) beef production from calves out of milk production raised on an organic grazing system is much more efficient than beef production from suckler systems. (Nguyen et al., 2010) investigated the environmental profile of a beef cattle system and three systems based on dairy bull calves for beef production. The CO_2 emissions related to land-use changes initiated by the use of concentrate, such as soybean meal, have been taken into account. It was found that the carbon footprint of bull calves from the dairy systems is similar to that of calves in the beef cattle system and relatively small differences occurred among the dairy-based systems. When calculating future opportunity costs, it needs to be taken into account whether land areas represent an opportunity to produce other foods or not. Grassland with no alternatives for agricultural production, or grassland that from a societal point of view is dedicated to maintaining a particular cultural landscape, should not be burdened with opportunity costs (Nguyen et al., 2010). An absolute reduction of GHG emissions can only be achieved if beef consumption decreases significantly.

2.2 MONOGASTRICS

GHGs from the pig sector account for about 15 % of livestock emissions, whereas the poultry sector contributes around 6 % (Lesschen et al., 2011; Philippe and Nicks, 2013). Poultry (laying hens and broilers) thus accounts for only a low proportion of GHG emissions in Europe (Figure 5); it follows that a further emissions reduction is less urgent here than in other livestock species. Pig production, in contrast, is an important contributor to polluting gases emissions like NH_3 and greenhouse gases (GHG). Apart from environmental aspects, animal welfare is also an issue of growing importance. In pig and poultry husbandry, most CH_4 originates from manures. The main sources of N_2O are: nitrogen fertilisers, land-applied animal manure, and urine deposited by animals with access to outside areas. Most effective mitigation strategies for CH_4 comprise a source approach, i.e. changing animals' diets towards greater efficiencies or optimised manure management (Monteny et al., 2006).

2.2.1 PRODUCTION SYSTEMS

A large part of European pig production is carried out in intensive systems with animals confined indoors being fed optimised diets. There are large variations in NH_3 abatement from the use of slatted floors and different bedding systems or a combination of both. A high proportion of the feed is imported rather than grown on the farm. Many large pig farms also do not have sufficient land for the utilisation of manure and depend on export of slurry. Although the biological productivity of these systems is often high, the externalities in terms of reduced animal welfare and environmental impact through losses of nutrients have been questioned. Organic pig production has emerged as an alternative with the multiple aims of improving animal welfare by supporting to a higher extent the pig's natural behaviour (Hermansen, 2003), and improve soil fertility by better linking crop and livestock production from an agro-ecological point of view. The differences between organic and conventional pig production are more fundamental than for example differences between dairy production systems, which may be why the percentage of organic pig herds is considerably lower than the percentage of organic dairy herds in Europe (Halberg et al., 2010).

Organic rules on grazing and access to outdoor area in pig production can be met in different ways. Compromises between considerations of animal welfare, feed self-reliance and negative environmental impact have to be made. According to Halberg et al. (2010) sows in Denmark are normally kept in huts on grassland and finishing pigs are raised in stables with access to an outdoor pen. One alternative practised is to also rear the fattening pigs on grassland all year round. A third method is a one-unit pen system mainly consisting of a deep litter area under a climate tent with restricted access to a grazing area. The GHG emissions of all free-range systems are significantly higher than those of the indoor fattening system – by 7 % to 22 %. When, due to the integration of grass-clover, carbon sequestration is included in the life-cycle assessment, the organic systems have lower greenhouse gas emissions compared to conventional pig production. Halberg et al. (2010) conclude that all free-range systems, especially the tent grass-clover system, have agro-ecological advantages over the indoor fattening system but are difficult to implement in practice due to leaching problems. Only if forage can contribute a larger proportion of pig feed uptake may the free range system be economically and environmentally competitive. Outdoor pig farming has expanded in recent years with around 45 % of the UK breeding herd kept outdoors in 2010 (Webb et al., 2014). In the UK, outdoor pigs are usually included in arable rotations. The magnitude of losses during and after pig production is likely to be influenced by soil type and condition, subsequent site management, vegetation and climate.

2.2.2 HOUSING

Animal welfare and environmental protection are increasingly important. Housing systems must be found that offer animal welfare while minimising the overall emissions of NH_3 and greenhouse gases. Compared with slatted-floor systems, litter systems in pig production present advantages in terms of animal welfare improvements, odour nuisance reduction and a better perception by consumers and neighbours (Philippe et al., 2013; Philippe and Nicks, 2013). The most frequent substrates are straw and sawdust. Compared to straw litters, sawdust litters produce less NH_3 and CH_4 but more N_2O. Increasing the amount of substrate also impacts emissions: typically it reduces NH_3 and N_2O but has variable effects on CH_4 production (Philippe and Nicks, 2013).The straw flow system is an animal-friendly housing system for fattening pigs, which can be operated economically on commercial farms (Amon et al., 2007). The fattening of pigs on deep litter bedded systems is more expensive and requests more labour than on slatted floor systems. The use of straw flow rather than straw deep litter could be a good compromise because of a reduced need for surface area, straw, labour and manure storage, combined with satisfying animal welfare. The straw flow system is associated with increased NH_3 emissions (+10%) but reduced GHG emissions (–55% N_2O, –46% CH_4 and –10% CO_2). Keeping group-housed gestating sows on partly straw bedded floor with permanent access to the concrete feeding stalls compared with fully straw bedded floor did not influence animal performance and NH_3 emissions and decreased CO_2 emissions by 40%. This CO_2 reduction was achieved despite a major decrease in N_2O emissions (–49%) (Philippe et al., 2013). The type of stable most commonly used by full-time producers in Denmark (Halberg et al., 2010) is a system with deep litter in the entire indoor area or deep litter / straw bed in half the area. The outdoor run consists of a concrete area from which the manure can be collected, as a way to comply with the environmental regulations aiming at preventing leaching. Research shows that very good production results can be obtained in such systems in terms of litter size, daily gain, feed consumption, and health (Hermansen, 2003). Use of straw in pig housing will increase due to concern for the welfare of the animals. In conjunction with naturally ventilated housing systems, straw allows the animals to self-regulate their temperature with less ventilation and heating (United-Nations, 2014).

Air inlets or outlets located near the manure surface increase the emissions due to a higher air exchange rate at interface (Philippe and Nicks, 2013). Using the minimum ventilation rate consistent with animal welfare can reduce odour emission rates. The ambient parameters must primarily respect the comfort of the animals. Moreover, the climatic conditions may alter pig behaviour, with indirect effects on emissions. The installation of water sprinklers to cool the animals or sufficient space could prevent increasing NH_3 emissions. House designs have to respect the natural excretory / lying behaviour of the pig to limit emissions. In systems with litter, the pen is sometimes divided into solid areas with litter and slatted dunging areas. Especially under warm conditions, pigs do not always use these areas in the desired way, using the littered area to dung and the slatted area to cool off in warm weather. Generally, pens should be designed to accommodate the desired excreting behaviour of pigs to minimise soiling of solid floors. This is more difficult in regions with a warm climate (United-Nations, 2014).

Frequent manure removal with a scraper is important to diminish the emissions within the building. The V-shaped scraper system is effective in reducing emissions since it is associated with separation of urine from faeces. Pit flushing is also an efficient method. Significant reduction by 45% for NH_3 and 49% for CH_4 were observed with this technique (Philippe and Nicks, 2013). Frequent flushing of slurry (normally once in the morning and once in the evening) causes nuisance odour events. Flushing slurry also consumes energy unless passive systems are used. The use of litter can reduce NH_3 emissions compared with liquid systems, depending on stocking density, the type of pig and the amount of litter (Webb et al., 2014). Emissions from litter depend greatly on particular conditions inside the manure (C/N ratio, aeration, temperature) (Philippe and Nicks, 2013).

Besides regulations on use of feedstuffs, organic pig production faces a major challenge in the regulations on housing. The sows need access to grazing in the summer time, and growing pigs need as a minimum requirement access to an outdoor run. In addition, the area requirements for indoor housing are higher than for conventional production. According to (Halberg et al., 2010) efforts to improve organic pig production should focus on the integration of livestock production and land use, while also considering environmental impacts on local and global scales.

Several mitigation techniques are available to reduce NH_3, N_2O and CH_4 emissions from pig houses, whatever the floor type. However, some strategies show contradictory effects depending on the circumstances and the gas. The choice of housing system is also guided by other factors, such as animal health, performance and welfare, agronomical values of manure, and clearly the investment and operating costs (Philippe and Nicks, 2013).

2.2.3 FEEDING STRATEGIES

Many odorous compounds are intermediate or end products of protein degradation. Therefore, protein is an important dietary compound that could be altered to reduce odour emission. Diets generally contain more protein than the pigs' requirement. The main reason is that the amino acids (AA) composition of dietary protein from feeds does not match the animals' requirements. The dietary levels are formulated to supply the minimum level of the most essential and limiting AA. This results in a surplus of other AA in the diet. Usually a large part of dietary protein is excreted via urine and faeces. Reducing protein or nitrogen (N) concentration in excreta decreases the availability of substrates that microbes can metabolise to odorous compounds. Le et al. (2007) found that feeding a diet more closely meeting the protein / AA

2.2.4 RECOMMENDATIONS FOR GOOD MANAGEMENT PRACTICES

NH_3 emissions from pig housing systems can be reduced by the following management practices without violating animal welfare and health:

- All free-range systems, especially the tent or hut grass-clover system, have agro-ecological advantages over the indoor fattening system.
- Applying phase feeding. Using regionally produced feed including grain legumes, such as soya. Increasing the non-starch polysaccharide content of the feed.
- Lowering CP of diets (2-3 %) is an effective strategy for decreasing NH_3 loss.
- Pens should be designed by locating the feeding and watering facilities to fit desired excreting behaviour of pigs and to minimise soiling of solid floors. Decreasing the surface area fouled by manure (inclined, smooth floors; slurry surfaces in channels with sloped walls).
- Compared to slatted-floor systems, litter systems in pig production present advantages in terms of animal welfare and odour nuisance reduction.
- The use of straw flow rather than straw deep litter is a good compromise because of a reduced need for surface area, straw, labour and manure storage, combined with satisfying animal behaviour and welfare.
- Keeping group-housed gestating sows on partly straw bedded floor with permanent access to the concrete feeding (-40 % COe compared with fully straw bedded floor).
- Adopting optimal straw treatments (spreading) in housing (e.g. 8 kg / week / pig). The use of extra straw reduces NH_3 by reducing airflow across surfaces soiled by urine.
- Reducing airflow and temperature (except where manure is being dried) over the manure surface and frequent removal from the building. Reducing the pH: litter NH_3 release is negligible at litter pH below 7.
- Removing the slurry from the pit frequently to an external slurry store or by flushing systems and additional treatment, such as liquid / solid separation and pH reduction.
- Lowering indoor temperature and ventilation rate, to consider animal welfare. Installing water sprinklers to cool the animals or providing sufficient space prevents increasing NH_3 rates.
- Circulating groundwater in floating heat exchangers to cool the surface of the manure in the under-floor pit to at least 12° C.
- Treatment of exhaust air by air scrubbers or biotrickling filters.

2.2.5 FURTHER RESEARCH NEEDS

Improved models and data for full life-cycle assessment are needed which integrate the whole process including manure storage and spreading, and consider free-range systems, animal nutrition and welfare.

Development of efficient feeding systems with reduced CP concentrations is required.

There is a need for more specific data concerning air quality and GHG emissions for bedding systems for pig production.

No studies have been undertaken that compare the N leached from pigs raised outdoors versus that arising from the application of pig manure from an equal number of housed pigs.

Further work would be useful to determine the potential for using fermentable sources of carbohydrates, which can reduce the pH of excreta and so further reduce emissions.

The behavioural habits of pigs, e.g. their tendency to play with straw, need consideration in the effectiveness of the straw treatments and require further investigation.

Conclusions regarding the impact of pork or chicken versus impact of milk or eggs require additional comparative studies and further harmonisation of methodology.

requirement of the pigs reduced odour concentration and odour and NH_3 emission from pig manure. This can be achieved by reducing the CP content of the diet and supplementing the diet with essential AA (Le et al., 2007; Webb et al., 2014) without compromising on pig performance. Emissions of NH_3 decrease with decreases in the CP content of pig diets, at all stages of manure management (Webb et al., 2014). However, the production of isolated essential amino acids is also a resource consuming and environmentally relevant process (many procedures require high amounts of organic solvents), which has to be included in the GHG balances of monogastric production systems. In organic agriculture, so far the feeding of isolated amino acids is not allowed.

Diet modification, such as reducing nitrogen inputs by reducing dietary CP without negatively affecting performance, is meanwhile a proven method to reduce nitrogen excretion. Feeding measures in pig production include phase feeding, formulating diets based on digestible/available nutrients, using low-protein amino acid-supplemented diets, and feed additives/supplements. A CP reduction of at least 2%-3% in the feed can be achieved, depending on pig production category and the starting point. It has been shown that a decrease of 1% CP in the diet of finishing pigs results in a 10% lower total ammoniacal nitrogen (TAN) content of the pig slurry and 10% lower NH_3 emissions (United-Nations, 2014). Pig production systems often depend largely on concentrated feed imported from outside the farm (van der Werf et al., 2005). The environmental burdens associated with the production and delivery of pig feed can be decreased by using more locally produced feed ingredients (including grain legumes, such as soya), so that transport is reduced, and using wheat-based diets rather than maize-based diets.

For poultry, the potential for reducing N excretion through feeding measures is more limited than for pigs because the conversion efficiency currently achieved on average is already high and the variability within a flock of birds is greater. A CP reduction of 1%-2% may be achieved depending on the species and the starting point (United-Nations, 2014).

3. MANURE MANAGEMENT

The treatment of manure influences the emission of NH_3, N_2O and CH_4 during the process of storage / composting / digestion but also influences biogeochemical processes in the soil.

Manure management refers to all activities, decisions and components used to handle, store and dispose of faces and urine from livestock with the goal of preserving and recycling the nutrients in the livestock production system IPCC, 2006a (Montes et al., 2013). This includes manure accumulation and collection in buildings, storage, processing, and application to agricultural land (Montes et al., 2013). Animal manure is a nutrient resource containing most of the essential elements required for plant growth and can be a significant source of N in both intensive and subsistence farming systems (Montes et al., 2013). Application of manure to agricultural land has manifold benefits as it maintains and improves soil quality, such as soil organic matter (SOM), the soil microbiota, water-holding capacity and increases crop yields (Diacono and Montemurro, 2010). Recycling of on-farm nutrients including animal manure on agricultural land is a key principle in organic and low external input farming systems. It aims at closing nutrient cycles at farm level and contributes to SOM reproduction at the same time (Gattinger et al., 2013, 2012; Leithold et al., 2014). Despite the benefits of animal manure, its management poses a substantial risk to the environment due to the gaseous losses of NH_3 and N_2O (Pardo et al., 2014; Sutton et al., 2011).

3.1 STORAGE AND HANDLING

Animal manures can release significant amounts of NH_3, N_2O and CH_4 during storage and are one of the major sources for agricultural greenhouse gases globally with an estimated global warming potential of 413 Mt CO_2 eq (Smith et al., 2008). There are several ways of managing animal manure, which can either be stored in liquid (e.g. slurry) or solid form (e.g. farmyard manure). Generally, intensive livestock systems use liquid manure management due to the large quantity of manure produced and the method of collection (Reid et al., 2004). CH_4 is mainly produced in strictly anaerobic environments, through the microbial decomposition of easily degradable organic compounds, whereas NH_3 and N_2O is usually associated with zones within the manure heap or slurry tank where an oxygen (O_2) gradient occurs as a result of nitrification–denitrification processes. NH_3 emission has been identified as the main pathway of N loss, accounting for up to 70 % in cattle manure (Montes et al., 2013) during these processes. It is of major concern because its subsequent deposition may disturb natural ecosystems through soil acidification and eutrophication of water bodies (Pardo et al., 2014). Apart from that, it has an indirect contribution to global warming since N deposited on soils and surface waters enhances N_2O formation (Smith et al., 2008). The formation of certain amounts of NH_3, N_2O and CH_4 seems to be unavoidable due to the inherent biological processes and the heterogeneous nature of waste piles and slurry facilities. However, the selection of management conditions plays a key role determining the magnitude of these emissions (Chadwick et al., 2011; Novak and Fiorelli, 2010).

Pardo et al. (2014) quantified the response of GHG emissions, NH_3 emissions and total N losses to different solid waste management strategies (conventional solid storage, turned composting, forced aerated composting, covering, compaction, addition / substitution of bulking agents and the use of additives). It emerged that improving the structure of the manure heap via addition or substitution of certain bulking agents significantly reduced N_2O and CH_4 emissions by 53 % and 71 %, respectively. Turned composting systems, unlike forced aerated composted systems, showed potential for reducing GHGs (N_2O: 50 % and CH_4: 71 %). Bulking agents (e.g. straw or woody materials to adjust C/N ratio) and both composting systems involved a certain degree of pollution swapping as they significantly promoted NH_3 emissions by 35 %, 54 % and 121 % for bulking agents, turned and forced aerated composting, respectively. Strategies based on the restriction of O_2 supply, such as covering or compaction, did not show significant effects on reducing GHGs but substantially decreased NH_3 emissions by 61 % and 54 % for covering and compaction, respectively. The use of specific additives (phosphogypsum, ferric chloride, aluminum sulphate among others) significantly reduced NH_3 losses by 69 %.

A quantitative evaluation / meta-analysis on the effectiveness of techniques to reduce gaseous N losses from liquid animal manure storage and handling is lacking until now. Certainly, the anaerobic digestion of slurry to biogas is effective in reduction of CH_4 and may also help to reduce N losses (see next chapter). A direct way to reduce gaseous N losses is to shorten the time manure is stored (Philippe et al., 2007). Furthermore, temperature is a critical factor regulating processes leading to NH_3 and CH_4 emissions from stored manure (Montes et al., 2013). Decreasing manure temperature to < 10°C by removing the manure from the building and storing under cold conditions can reduce GHG and NH_3 emissions (Montes et al., 2013; Smith et al., 2008). More than 90 % of the non-CO_2 GHG emissions from untreated slurry originate from CH_4 emissions during slurry storage. This means that GHG abatement measures in slurry management are most effective if they reduce CH_4 emissions during slurry storage (Amon et al., 2006). Aeration of cattle and pig slurry increased NH_3 and N_2O emissions by 86 % and 127 %, respectively, but decreased CH_4 by 57 % (Amon et al., 2006). Several types of liquid manure covers have been reported in the literature such as natural crusts on slurry manure stored with high solids content such as: straw or wood chips; oil layers; expanded clay pellets; wood; semipermeable and sealed plastic covers. However, they often do not reduce NH_3, N_2O and CH_4 at the same time. Semipermeable covers for instance are valuable for reducing NH_3, CH_4 and odour emissions, but they often increase N_2O emissions (Montes et al., 2013). Amon et al. (2006) reported an increase of all three gases when a straw cover was placed on the surface of cattle slurry.

	CO$_2$-C (%)			CH$_4$-C (%)			N$_2$O-N (%)			NH$_3$-N (%)			Total N (%)		
	N	Mean	SD	*N*	Mean	SD	*N*	Mean	SD	*N*	Mean	SD	*N*	Mean	SD
Waste type															
Cattle manure	27	40.0%	14.3%	23	3.2%	2.7%	29	1.3%	1.5%	40	11.6%	14.6%	38	27.4%	15.7%
Dairy manure	19	34.8%	14.4%	26	0.9%	0.9%	29	0.6%	0.8%	20	9.4%	7.7%	16	23.9%	14.3%
Pig manure	69	48.0%	15.6%	48	1.5%	2.3%	60	2.7%	2.2%	81	17.1%	12.5%	94	39.4%	17.6%
Poultry manure	18	42.3%	12.6%	4	0.1%	0.1%	13	1.3%	2.3%	38	16.7%	14.3%	37	35.8%	21.9%
Green waste	6	55.7%	17.4%	2	1.4%	0.4%	2	1.0%	0.7%	6	11.2%	9.4%	2	36.3%	40.0%
Treatment type															
Storage	40	40.9%	12.9%	37	1.1%	1.9%	51	1.5%	1.8%	70	12.5%	12.4%	73	35.7%	18.4%
Turned	56	51.4%	15.9%	36	1.9%	2.1%	39	1.2%	1.3%	44	21.0%	16.6%	57	44.6%	17.6%
Forced aeration	36	50.0%	21.4%	6	0.3%	0.4%	7	1.2%	0.8%	38	18.8%	18.1%	31	39.7%	19.9%
F. aeration+Turned	41	36.3%	12.4%	17	3.2%	3.1%	28	3.8%	2.3%	40	16.6%	9.0%	44	33.3%	17.9%
Covered	4	25.0%	15.8%	7	0.9%	1.4%	7	1.5%	1.4%	9	5.9%	7.1%	14	16.7%	9.2%
Compacted	7	24.5%	10.7%	4	3.0%	4.5%	7	0.6%	0.7%	7	6.4%	7.1%	5	20.4%	12.1%
Temperature															
Cool temperate	33	37.4%	15.0%	46	0.7%	1.1%	59	1.3%	1.4%	87	12.4%	12.3%	57	26.4%	18.8%
Warm temperate	137	44.1%	16.9%	54	2.4%	2.8%	73	2.3%	2.3%	109	16.5%	13.9%	144	37.8%	17.2%
Annual rainfall rate															
Dry	46	44.4%	11.3%	39	2.7%	2.8%	39	2.0%	2.1%	36	21.2%	10.6%	46	26.6%	13.3%
Moist	128	45.4%	18.7%	65	1.2%	1.9%	106	1.8%	2.1%	181	14.3%	14.4%	177	36.7%	19.5%
Wet	12	22.0%	2.0%	0	-	-	0	-	-	0	-	-	12	55.3%	12.5%
Duration															
<1 month	31	38.5%	21.9%	7	1.6%	3.7%	6	2.1%	2.5%	43	14.0%	11.0%	48	38.2%	20.1%
1-3 months	85	42.3%	14.4%	45	1.7%	2.4%	66	2.0%	1.9%	75	18.4%	14.2%	84	32.5%	16.2%
>3 months	70	47.6%	17.9%	57	1.7%	2.1%	73	1.7%	2.2%	99	13.9%	14.9%	103	37.1%	20.5%
Scale															
Commercial	92	42.2%	15.2%	75	1.6%	2.2%	96	1.3%	1.7%	124	14.8%	14.6%	127	34.7%	19.0%
Pilot	92	45.6%	19.3%	32	2.0%	2.7%	43	2.9%	2.2%	84	17.1%	13.8%	97	38.8%	18.7%

Table 1: Number of observations (N), mean and standard deviation (SD) of cumulative gaseous emissions for some of the factors with a potential influence on C and N losses from management of solid animal manure (adapted from Pardo et al. 2014)

3.2 BIOGAS PRODUCTION

Most CH_4 is produced during manure storage. Therefore, reducing storage time, lowering manure temperature by storing it outside during colder seasons, and capturing and combusting the CH_4 produced during storage are effective practices to reduce CH_4 emissions. Anaerobic digestion with combustion of the gas produced is effective in reducing CH_4 emissions and the organic C content of manure; this increases readily available C and N for microbial processes while creating little CH_4 and increased N_2O emissions following land application (Montes et al., 2013).

Use of biogas digestion may reduce N-losses and lead to higher farmland productivity caused by improved plant availability (Clemens et al., 2006; Montes et al., 2013; Novak and Fiorelli, 2010) compared to stored liquid manure. Digestion is linked to losses of organic C in manure but also to the production of energy (Möller, 2009; Möller et al., 2008; Montes et al., 2013). In a biogas plant the manure will be processed in a closed system. PH and the NH_3 concentration in the manure will be enhanced. Even after completion of digestion, the manure still releases NH_3 and CH_4 and therefore requires post-fermentation storage facilities to capture these losses. In general, the fermented substrates have to be stored in closed chambers to reduce the losses of CH_4 and NH_3 (Möller, 2009). The higher pH and the higher NH_3 concentration in the manure increase the potential for NH_3 losses during field application (Kim et al., 2013). The aerobic post-treatment of digested and separated solid manure, a treatment to improve its fertiliser quality, raises the emission of CO_2 eq two- to three-fold compared to composting (Cuhls et al., 2011).

However, if the biogas plant is not hermetically closed, CH_4 will be lost by leakage to the atmosphere. The potential leakage of the plants should be checked regularly during operation (Cuhls et al., 2011; Jensen et al., 2012). CH_4 emissions reduce the effect of GHG mitigation by biogas production.

3.3 RECOMMENDATIONS FOR GOOD MANAGEMENT PRACTICES

- Composting of solid animal manure is an effective means of reducing GHG emissions.

- The combination of practices for handling solid manure, such as composting, coverage of the piles and the usage of specific additives appears be the best option for reducing NH_3, N_2O and CH_4 at the same time.

- Digestion of liquid manure in biogas fermentation plants with covered storage capacities is even more effective than composting and reduces GHG emissions before application. However, it needs appropriate application techniques and an adaption of application in quantity and in time to reduce the GHG emissions of the whole application.

- Anaerobic fermentation of manure for biogas production contributes to the replacement of fossil fuels by using the obtained energy for heating and/or as electricity.

- Appropriate application of manures in the field to reduce gaseous losses. This is of particular importance for slurries rich in NH_3.

3.4 FURTHER RESEARCH NEEDS

There are many different systems with excellent opportunities to reduce emission losses with low losses, but with negative influences on the next step of the fertiliser chain. For further research, it is necessary to get an overview of the whole fertiliser chain from the animal nutrition to manure, to soil and plants.

In further research, experiments should include the whole chain of different steps from animal nutrition to manure to soil and plants. Over the whole farming system the GHG emissions and the nutrient cycle should be calculated using results from existing research to gain an overview of the best combinations of the different steps.

4. CROP AND FORAGE PRODUCTION

The soil nitrogen cycle is a complex interaction of different processes that are governed by bio-chemical, microbial and plant interaction. In times of excess nitrogen, e.g. after fertilisation or tillage, the risk of N losses is high. Whereas NO_3 and NH_3 losses are important in terms of yield reductions and large-scale environmental problems, N_2O emissions play an important role in climate change. Analyses comparing N fluxes in different farming systems showed a higher N efficiency in organic crop rotations when taking into account the changes in soil organic nitrogen stocks (Küstermann et al., 2010). N_2O emissions following land application occur as a by-product of nitrification and denitrification processes. These microbial processes depend on temperature, moisture content, availability of easily degradable organic C, and oxidation status of the environment, which make N_2O emissions and mitigation results highly variable (Li et al., 2013; Montes et al., 2013; Novak and Fiorelli, 2010).

4.1 FERTILISER APPLICATION AND OTHER AMENDMENTS

The timing and rate of fertilisation is often mentioned in connection with N reduction aims. There is the general recommendation that application should be performed at the time of the greatest crop need and that rates should be adjusted to avoid excess nitrogen in soils (Gerber et al., 2013; Smith, 2010). NH_3 is lost directly out of NH_4 fertilisers. This is an important issue especially regarding slurries. NO_3 can be lost directly from NO_3 fertilisers or when microorganisms transform fertiliser NH_4 into NO_3. N_2O losses are more complex, as different microbial processes lead to N_2O production. Optimising techniques to reduce emissions from one process can stimulate N_2O production by another. Furthermore, reducing NO_3 and NH_4 losses increases the amount of N available in the soil and consequently the risk of N_2O emissions (Paulsen et al., 2013). Regarding N_2O, the highest losses can be expected under soil conditions that stimulate microbial activity (warm and moist) (Paulsen et al., 2013). Adjusting the timing and rate of fertiliser applications is a difficult task as crop needs and soil conditions depend on the prevalent and unpredictable weather conditions (Paulsen et al., 2013). It is also important to assure that crops are sufficiently supplied by other nutrients, as a nutrient deficiency can limit N uptake (Paulsen et al., 2013).

4.1.1 APPLICATION OF SYNTHETICALLY MANUFACTURED FERTILISERS

For synthetically produced fertilisers, peak emissions of N_2O were reported for the first two weeks after fertilisation. The emission quantity is variable, depending on the prevailing weather conditions (Hyde et al., 2006). Fertilisers containing NO_3 are more susceptible to N_2O losses than urea or NH_4-based fertilisers (Schils et al., 2013). N_2O emissions have been found to increase non-linearly with N-application rate. That means that fertilising above the crop demand leads to an exponential increase in N_2O emissions (Kim et al., 2013; Snyder et al., 2009). To find an environmental optimum fertiliser rate, yields and N losses should both be considered in the farm N budget (Tuomisto et al., 2012). Placing fertilisers more than 5 cm deep into the soil has been found to reduce N_2O emissions in reduced and no-till systems under humid conditions (van Kessel et al., 2013), which is only possible for liquid fertilisers. (Paulsen et al., 2013) however reported variable results for deep placement and banded application for N_2O, while both techniques can reduce NO_3 and NH_3 emissions. Splitting the yearly N rate to several applications with lower N amounts during the period of highest crop need has often been reported to be efficient in reducing N losses (Smith, 2010; Snyder et al., 2009). Slow-release fertilisers have the potential to reduce N_2O emissions (Li et al., 2013; Snyder et al., 2009) but their efficiency depends on environmental conditions than can desynchronise N release and plant root uptake (Paulsen et al., 2013).

4.1.2 APPLICATION OF ORGANIC FERTILISERS

While straw-based solid manure can emit less NH_3 than slurry after surface spreading on fields, slurry provides a greater opportunity for reduced-emissions applications (Powell et al., 2008). Liquid animal slurry contains mostly directly available NH_4 that can be lost immediately by NH_3 volatilisation or as N_2O after application. Those peak emissions are rather short, mostly less than a week (Carozzi et al., 2013). The N losses by NH_3 can be as high as 10-40 % (United Nations, 2014). This is both economically and environmentally of interest. Abating NH_3 was therefore extensively investigated and control measures summarised e.g. in the EU ECE/EB.AIR/120 report (United Nations, 2014). Measures include a switch in application techniques from surface application to band spreading by trailing hoses (reduction by 30-35 %) or trailing shoes (30-

60 %), and injection techniques differing between open-slot injection (70 %) or closed-slot injection (80-90%). A direct incorporation of slurries can also reduce NH_3 emissions by 30-90 % depending on inversion / non-inversion and depth of tillage. Besides NH_3, CH_4 emissions by slurries are also reduced after quick incorporation (Montes et al., 2013). Injection works well when it is combined with an-aerobic digestion and solids separation by improving infiltration. Additives such as urease and nitrification inhibitors can also be added at this stage (Montes et al., 2013). Their effects are described below. However, reducing the loss of NH_3 increases the amount of available nitrogen in soils. Higher N_2O emissions were consequently reported when injecting slurry (Möller and Stinner, 2009; Thangarajan et al., 2013) or when slurry was tilled into the soil (Olesen et al., 2006). The impact of application techniques on N_2O emissions depends on the prevalent climatic conditions (Li et al., 2013). Slurry injection can in addition induce local zones of anoxic properties in soils which may further promote denitrification and even methanogenesis (Gerber et al., 2013). Due to the high risk of N losses, it has been recommended not to apply slurry in late autumn / early winter (Tuomisto et al., 2012) or during the warmest part of the day (Novak and Fiorelli, 2010). Any evaluation of the climatic relevance of the measured gas emissions from the different application techniques has to compare all GHGs. It is evident that NH_3 emission reduction, which may be achieved with injection, can be at least compensated by increased N_2O emissions. The injection system needs, in addition, the highest tractive forces. The results indicate that on arable land, trail hose application with immediate shallow incorporation, and on grassland, trail shoe application, bear the smallest risks of high greenhouse gas emissions when fertilising with co-fermented slurry (Judd et al., 1999).

For solid manures, NH_3 reduction of 30-90 % through direct incorporation within a day after application has been suggested (United Nations, 2014). Applied on surface, NH_3 emissions varied with manure type while CH_4 and N_2O emissions have been found to be negligible (Pinares-Patino et al., 2003). Ploughing solid manures, however, stimulated N_2O fluxes by mineralising the manures in addition to the soil organic matter (Olesen et al., 2009). A problem in terms of a balanced distribution of nutrients within a farm can arise through manure separation. The liquid fraction contains mainly NH_4 and K, the solid fraction more P and organic nitrogen. The liquid fraction is transported only at smaller distances because of its weight, whereas the solid fraction can also be applied to fields at longer distances to the farm. This unbalanced nutrient distribution can provoke a surplus of various nutrients in fields near the farm premises, causing additional problems (Herrmann, 2013; Hutchinson et al., 2007). As animals and biogas plants produce effluents continuously and plant nutrient demand is seasonally constrained, large storage capacities are needed to provide fertilisers at the optimal time (Amon et al., 2004). Green waste compost application was reported not to increase N_2O emissions (Vaughan et al., 2014), or to do so only in a wet season (Ball et al., 2014). Green manures and crop residues taken from the field for biogas digestion decreased soil N contents. However, applying the digestate in the field induced higher NH_3 emissions compared to undigested slurry. Liquid effluents from the biogas digester also increased N_2O emissions (Stinner et al., 2008).

4.1.3 UREASE AND NITRIFICATION INHIBITORS

Urease inhibitors have been recommended to reduce NH_3 emissions after urea application (Chadwick et al., 2011) and nitrification inhibitors (NI) have been shown to reduce N_2O after synthetic and organic fertilisation. Efficiency was reported to decrease in the order: grassland, oxic arable and paddy soils (Akiyama et al., 2010). In grazed pastures, the application of dicyandi-amide (DCD) as nitrification inhibitor highly reduced NO_3 and N_2O emissions from urine patches (Ball et al., 2012). In combination with urease inhibitors an even greater effect was measured (Mutegi et al., 2010; Schils et al., 2013). However, the persistence of NI's in grazed pastures plays a great role in their overall effect. For DCD a half-life of 85 days (< 10° C) and 50 days (15° C) was reported in English pastures (Tuomisto et al., 2012). Applying NI's, NH_4 is more prevalent in soils and prone to loss than NH_3, especially in soils with a high pH and low CEC (Kim et al., 2013) or might have a priming effect, contributing to the mineralisation of soil organic matter (Luo et al., 2010). As NI's do not necessarily increase yields and are thus more an environmental measure, political incentives are needed to make them economically attractive to farmers (Li et al., 2013). Side effects of nitrification inhibitors on soil organic matter and soil biota are not well known.

4.1.4 BIOCHAR

Biochar, which is charred organic matter from woody or other plant material, is considered to have positive effects on overall soil fertility and has been reported to decrease N_2O emissions in soils. The reduction potential seems to depend on feedstock materials, pyrolysis conditions, soil texture and the type of N fertiliser (Cayuela et al., 2014). There are indications that hydrochars produced during hydrothermal carbonisation instead of pyrolysis do not reduce N_2O emissions (Kammann et al., 2012). N_2O emissions from urine patches were also reduced when incorporating biochar into pasture soils. The persistence of the observed effect however, remained unclear (Saggar et al., 2004).

4.2 ARABLE FARMING

4.2.1 CROP ROTATIONS INCLUDING TILLAGE

Crop rotations include different phases of crop and fallow periods, tillage, inter crops and, in organic farming, also ley periods with perennial crops (mostly grass and legume species). Next to fertilisation, tillage is often reported to pose a great risk of N loss. When tilling the soil, microbial activity is stimulated and soil organic matter is mineralised. NO_3 leaching and N_2O emissions can be greatly increased, especially as no plants are available to take up the excess nitrogen immediately. So it is recommended that soils should be covered with plants as soon as possible after tillage. Bare fallows in winter are not recommended. Green manures as catch crops take up soil nitrogen and decrease the risk of N losses. As mulching and incorporation of green manures can have a similar effect as fertilisation, considerable N losses can be expected depending on the time of incorporation and type of green manure (legume, non-legume). Tilling a winter radish cover crop e.g. increased N_2O more when ploughed than when less intensively tilled in spring (Velthof et al., 1996). Crop residues can have an effect similar to green manures when tilled. A high C/N ratio can immobilise soil N whereas a low C/N ratio stimulates mineralisation and N is prone to be lost (Chen et al., 2013; Novak and Fiorelli, 2010). Leaving crop residues on surface induced less N_2O emissions than incorporation by ploughing in autumn (Novak and Fiorelli, 2010). Tillage intensity is a well-studied area in terms of N_2O. Van Kessel et al. (2013) summarised in a meta study that no differences between ploughing (CT) and reduced tillage (RT)/no-tillage (NT) were found in humid climates. They further found an increase in the first years and decrease of N_2O emissions after more than 10 years of conversion to RT/NT in dry climates. A recent study by Ball et al. (2014) also found no differences between tillage intensities on GHG emissions in an organic crop rotation. Impacts of tillage intensity on CH_4 emissions were found to be of minor importance (Shan and Yan, 2013). The destruction of perennial leys through tillage, which are characteristic of organic arable rotations, has a large effect. They serve forage production, break pest and disease cycles and introduce N through biological nitrogen fixation when legumes are grown. During the ley phase, N_2O emissions have been found to be lower than in N fertilised systems (Jensen et al., 2012; Schmeer et al., 2014). Cutting a legume-grass ley and removing herbage induced less N_2O emissions than mulching (Yamulki and Jarvis, 2002). Biologically fixed nitrogen can have a similar effect on annual N_2O emissions when ploughed and released compared to fertiliser applications (Smith, 2010). Thus it is not surprising that several studies have reported high N_2O emissions after ploughing a ley (Ball et al., 2014; Ball et al., 2007; Yamulki and Jarvis, 2002). The optimum time for ley termination was suggested to be early spring, when the cold restricts nitrogen mineralisation at first but provides N for the subsequent crop later on (Ball et al., 2014). In humid climates this will be a difficult task as soils are wet in spring and tillage hardly possible. Overall, legume crops and legume-based pastures used 35-60% less fossil energy than N fertilised cereals or grassland due to the avoidance of N fertilisers (Jensen et al., 2012).

4.2.2 TECHNICAL INNOVATIONS

Site-specific fertilisation coupled with precision farming techniques present an opportunity to account for soil heterogeneity and different crop N demands within a field and therefore to reduce fertiliser amounts and adjacent N loss (Paulsen et al., 2013; Sehy et al., 2003). Furthermore, controlled traffic reduces the compacted area in arable land and grassland by fixed tracks. Compaction has been shown to increase N_2O significantly (Ruser et al., 1998; Schmeer et al., 2014).

4.3 GRASSLAND AND PASTURE MANAGEMENT

Permanent grassland can either be managed by cutting for forage production, or as pastures for grazing or as mixed forms. When ungrazed, it was has been reported that N_2O emissions from unfertilised legume based grassland were lower than from fertilised grassland with no legumes (Li et al., 2013). This can be attributed to negligible direct N_2O emissions found for the process of biological nitrogen fixation (Li et al., 2013; Tuomisto et al., 2012). NO_3 based fertilisers produced more N_2O than NH_4 based fertilisers when applied to actively growing crops (Smith, 2010). As soon as pastures were grazed, N_2O emissions from urine patches were a large and very heterogeneous source of N_2O (Luo et al., 2010). Oenema et al. (2006) found a loss of 0.1-3.8 % of urine-N and 0.1-0.7 % of dung-N from patches as N_2O. The difference in N_2O emissions between legume based pastures (unfertilised) and grass based (fertilised) pastures mentioned above was offset by grazing (Li et al., 2013). Concerning NH_3, the rapid infiltration of urine from patches into the soil was found to restrict emissions. Thus the NH_3 loss per animal was less for grazing than from housed animals whose excreta are stored and applied to the field later on (United Nations, 2014). Delaying fertiliser application after grazing can reduce N_2O (Luo et al., 2010). Restricted grazing is also often reported to reduce patch derived N_2O emissions. This can either be a daily restriction (Gerber et al., 2013; Li et al., 2013) or a seasonal restriction avoiding wet soils in winter where hoof compaction favours anoxic zones leading to high N_2O and maybe CH_4 emissions in conjunction with urine (Smith, 2010). As plant N uptake is reduced in the cold season, collecting excreta during animal housing and spreading effluents in times of crop need improves overall N use efficiency (Li et al., 2013). Winter management, however, must be used with care to avoid trade-offs such as reducing N_2O emissions from paddocks, but increasing NH_3 loss from animal houses. Rotational grazing is a restriction in terms of space. Animals feed more efficiently on herbs, which was found to reduce CH_4 emissions per unit animal weight gain compared to continuous grazing (DeRamus et al., 2003). Pastures have the potential to store carbon when grazed (Novak and Fiorelli, 2010). Accounting N_2O and CH_4 emissions to assess the global warming potential (GWP) of grazed systems, extensively managed pastures in terms of stock density and fertilisation were found to reduce GWP (Allard et al., 2007) and the emission factors of N_2O (Flechard et al., 2007). A certain stocking density is recommended, however, to assure carbon sequestration by the continuous growing of forage plants (Janzen, 2011; Novak and Fiorelli, 2010). As pastures are often on organic soils in the form of wetlands or peatlands, waterlogging of these sites poses the risk of CH_4 and N_2O emissions. Drainage can have variable results depending on the height of the water table (Smith, 2010). Renovation of grasslands can increase soil N mineralisation and as a result N_2O emissions. Renovations in autumn produce higher N_2O emissions than in spring. Removal of grazing animals in the months before grassland renovation is suggested to reduce the potential for N_2O losses. Dairy production systems in some parts of Europe are based on ley–arable rotations. In the ley phase of such rotations, N accumulation occurs in soils not disturbed by tillage operations. Consequently, a considerable N surplus occurs in grasslands, particularly under grazing regimes, where a large part of the N in ingested grass is recycled to soil via urine and faeces (Ledgard et al., 2009). Grassland cultivation almost always results in a substantial residual effect and the mineralisation of N often exceeds the requirement of the succeeding crop. Thus, there is a high risk of N losses following sward cultivation. Management practices to control N losses, including N_2O emissions, comprise delayed ploughing until late winter or spring, the use of efficient catch crops after ploughing and a reduction in fertiliser N application to cereals after ploughing. Because grass pasture requires inputs of N fertiliser, this type of pasture will have additional fertiliser-specific losses. In an Australian study, N losses from total denitrification were significantly less from unfertilised clover / ryegrass pasture (Schils et al., 2013).

4.4 RECOMMENDATIONS FOR GOOD MANAGEMENT PRACTICES

- All fertiliser N sources should be included in the calculation of fertilisation balances. This means that organic fertilisers and legume fixed N should be accounted for, complemented by synthetic fertilisers as needed to meet crop demands.

- The application of liquid organic fertilisers should be made during the coldest part of the day using trailing hoses, shoes or injection techniques. Smaller amounts should be applied at several points in time when crop demand is high. Absorptive soils, windless conditions and following rain diminish NH_3 emissions. Solid manures and composts are best incorporated in the coldest part of the year.

- A large amount of synthetically produced N fertilisers can be substituted by biological nitrogen fixation (legumes), reducing CO_2 from fossil fuel consumption. Synthetic fertilisers are only needed when not enough organic fertilisers are available; synthetic fertilisers are not allowed in organic farming.

- Tillage intensity and frequency should be minimised as far as possible, adjusted to crop needs and performed in the coldest and driest soil conditions possible.

- Leys should best be tilled in the coldest and driest soil conditions possible, followed by a crop or catch crop that takes up the excess nitrogen rapidly.

- Green manures should replace bare fallow periods to take up soluble nitrogen forms.

- Site-specific fertilisation is an effective measure to reduce N losses.

- Controlled traffic both in arable and grassland are highly recommended to restrict soil compaction. Slurry should be spread with tubes to reduce traffic with heavy slurry tanks.

- Grazing should be restricted to dry soil conditions. Rotational grazing increases nitrogen use efficiency.

4.5 FURTHER RESEARCH NEEDS

N in organic farming comes from variable sources: organic fertilisers, green manures, leys where it is stored and released from the soil N pool. N losses are thus even more complex and difficult to predict because of diverse responses to weather and soil conditions. Monitoring N losses over whole crop rotations would give a clearer picture than results from single crops by year.

Dynamics of N losses (N_2O, NO_3) after green manure and ley incorporation depending on plant type and tillage timing/depth should be further investigated considering different soil types and climatic conditions.

The concept of site-specific fertilisation by precision farming approaches is only poorly developed for organic fertilisation and needs further attention to reduce N losses and improve profitability.

Biochar effects on soil N losses and practicability for farmers is a new and interesting approach, which should be further investigated.

The pre-crop effect on N_2O emissions has been studied poorly in the past and needs to be further investigated.

5. CONCLUSIONS

Global anthropogenic greenhouse gas (GHG) emissions are further increasing, driven largely by economic and population growth. The agriculture and forestry sector accounts for about a quarter of net anthropogenic GHG emissions – mainly from deforestation and agricultural emissions from livestock, cropland and nutrient management. Agriculture impacts both GHG emissions and global nitrogen cycles negatively, two main factors threatening the stability of the planet (Rockström et al., 2009). During the last years, global consumption of animal derived products (meat, milk, eggs) rose and is likely to increase further along with population growth and change in consumption patterns. The whole food production chain entails high environmental costs because of large GHG emissions from plant and livestock production, food waste along the supply chain and consumption behaviour.

Based on a review of the scientific literature we compared different livestock production systems, housings, feeding strategies, animal food production and manure management in terms of GHG and mitigation options, based on a life-cycle approach. To judge the appropriateness of mitigation options, it is essential to assess their impact on the carbon and nitrogen cycles at the whole farm level. However, this is far from easy as each mitigation option involves trade-offs between processes and the associated material fluxes, and there are often interactions and feedbacks among mitigation options. The choice of a set of mitigation options will therefore rely on careful assessment of the balance between their beneficial and adverse effects. It is important to consider the effects of mitigation measures on animal welfare and health. Animal welfare may be affected in several ways: e.g. through feeding regimes, housing conditions and grazing conditions. Another factor to take into consideration is the period needed to establish the balance between beneficial and adverse effects of mitigation options. GHG (especially N_2O) and NH_3 emissions vary greatly over time, particularly because of pedo-climatic conditions and biological reactions governing them. The effectiveness of a mitigation measure will thus exhibit strong year-to-year variations. Furthermore, the delay time lag between the implementation of a mitigation measure and real reduction of emissions will vary greatly among mitigation options. For instance, an improvement in manure application techniques will have an immediate effect on NH_3 emissions, whereas a change in land management or in soil tillage affecting the soil biogeochemical properties (and thus soil carbon sequestration and N_2O emissions) will probably involve a time scale of several years. Before its implementation, a mitigation option should therefore be assessed in the context of the whole farming system, or with an even broader scope by including upstream and downstream chains. This needs to be done at least on the time scale of the crop rotation and, if possible, in a manner considering other issues beyond global warming, e.g. water quality, soil fertility, animal welfare, biodiversity. The different mitigation options need to be carefully implemented, taking into account the circumstances of every single farm, depending on region, climate, soil conditions, socio-cultural aspects and regulations. The basis of all mitigation actions should be a sustainable agricultural system such as low external input or organic farming. The aim of sustainable agriculture is to establish environmentally sound production systems by limiting the adverse effects of agricultural activities on all the components of the environment. Non-commodity ecosystem services and incentives should be considered in all policy initiatives related to the goals of sustainable agriculture. Consumers should be encouraged to eat less animal-based food and policymakers should address food waste issues and establish reduction options.

Modelling approaches, combining biophysical and decisional simulations, would be useful to assess the balance of different sets of mitigation options by taking into consideration trade-offs, interaction and feedback among practices at farm level, and by evaluating their impact upon environmental and agronomic aspects in the various regions of the world. These will increase the ability to implement sustainable agro-ecological farm systems at local and regional scales. Furthermore, it is vital to address the social and economic context in both research and farm practice, as land use and land-use change is directly linked to human consumption patterns. Global perspectives tempt us to seek and advocate ubiquitous "best management practices" (BMP), but each hectare of land is unique, the sum of myriad interacting factors. What is needed is "place-based research", which recognises the distinctness of each local ecosystem and seeks the most appropriate system for conditions there. In any search for more enduring and resilient systems, the critical variable is time.

The following actions are recommended from the facts and figures discussed above for the various sections on food consumption, production and farming. We refrain from stating specific figures here. The potential for emissions reduction varies greatly depending on local conditions and more specific research data are needed. We rather follow first-order estimates when ranking the actions below in terms of their mitigation potential. If not specified otherwise, recommendations refer to the total of all GHG and NH_3 emissions.

FARM MANAGEMENT

1. Re-integrate livestock and plant production, either at farm or regional level, to generate mixed farming systems as has been the case for centuries and is still practised in organic farming. This could also be achieved through greater exchange of manure and forage between farms with and without livestock respectively.

2. It is possible to reduce NH_3 losses significantly, but this requires a whole-farm system approach to avoid negative trade-offs, because intervening in one part of the system may affect NH_3 losses in other parts. Therefore N balances at farmgate level, in which all N forms and other nutrients are considered for sound nutrient management, are recommended.

3. Optimise productivity and thus N use efficiency by improving farm management. The rate at which improved management techniques can be introduced depends on legislation, suitability in practice and their impacts on net production costs.

CROP PRODUCTION/MANURE MANAGEMENT

1. Ensure that N-fixing legumes are a significant proportion of crop rotations in order to reduce the persistence of reactive N forms, improve soil fertility and replace fertiliser production based on fossil fuels.

2. Recognise manure as a valuable resource rather than viewing it as waste material. Make manure recycling to croplands mandatory to end unnecessary disposal, in order to promote plant nutrition, soil organic matter reproduction (carbon sequestration) and soil fertility built-up.

3. Apply liquid organic fertilisers during the coldest part of the day using appropriate spreading techniques such as trailing hoses, shoes or injection techniques.

4. Minimise tillage intensity and frequency as far as possible.

5. Instead of fallow periods cover crops should be established to take up soluble nitrogen forms and serve as green manure.

LIVESTOCK PRODUCTION/MANURE MANAGEMENT

1. Improve animal health and nutrition, welfare, performance, fertility and therefore overall performance and longevity of livestock. This reduces the number of animals required and the associated GHG emissions.

2. Establish low-protein animal feeding and phase feeding by using locally produced feed, including grain legumes adapted to real needs.

3. Appropriate (climate, soil, and time) grazing management is a reliable emission reduction measure; the amount of reduction depends on the daily grazing time and the cleanliness of the house and holding area.

4. The use of straw flow rather than straw deep litter to replace slatted floors is a good compromise between production costs, animal welfare and NH_3 emissions. This reduces the need for surface area, straw, labour and manure storage, while also satisfying animal behaviour and welfare requirements.

5. Reducing airflow and temperature over the manure surface and ensure frequent manure and slurry removal from housing systems.

FOOD CONSUMPTION / PROCESSING / FOOD TRADE

Reduce food waste along the entire food chain and motivate people to consume less animal-based food.

6. POLICY RECOMMENDATIONS

Ambitious targets for ammonia and methane need to be established under the European Union's National Emissions Ceilings (NEC) Directive, on a progressive basis between 2020, 2025 and 2030, and made legally binding as part of Member States' Emissions Reduction Commitments (ERCs).

Targets under the NEC Directive should contribute to the EU's climate and energy policies. A new Effort Sharing Decision (ESD) under the EU's 2030 framework for climate and energy policies must target more ambitious non-CO_2 emission reductions for the agriculture sector – in particular for nitrous oxide emissions.

The European Commission and Member States need to ensure that the introduction of voluntary coupled support under CAP – especially for the livestock sector – does not cause negative impacts on emission objectives.

Organic farming and agrienvironment-climate measures under EU Rural Development Programmes can help to improve soil management, nutrient availability and nutrient use efficiency, and should be prioritised. This includes agro-ecological practices such as the recycling and application of organic fertilisers, reduced tillage, optimised crop rotations and the cultivation of nitrogen-fixing leguminous crops.

Rural development advisory services and knowledge transfer measures that stimulate the uptake of advanced agro-ecological approaches must be prioritised. Capital investments are needed to implement practices such as improved manure management and storage as well as better utilisation and recycling of nutrients from safe sources such as household and urban organic waste. Such practices contribute to closing nutrient cycles.

Sustainable biogas production should be promoted to reduce methane and ammonia emissions.

Targeted funding under the EU Research Framework Horizon 2020 and the European Innovation Partnership for Agriculture can help to improve ammonia, nitrous oxide and methane emission reductions through greater understanding of enhanced crop and live-stock management practices as well as the potential trade-offs. Stakeholder-led problem-solving through the establishment of operational groups orientated towards agro-ecological innovation should be prioritised. The organic concept for agriculture, food systems and food consumption can be used to address many relevant research and innovation factors. Knowledge gain and performance improvement triggered by research are over-proportionally high.

It must be recognised that one-size-fits-all models do not work. Assessments of the diversity of multi-functional agro-ecological systems such as organic farming are needed in order to ensure reliable comparative and quantifiable assessments of performance in terms of emissions reductions and to prevent biases towards unsustainable, but easily quantifiable solutions that would give rise to unnecessary trade-offs. Efforts undertaken by the agriculture sector to reduce non-CO_2 emissions must be delivered in a coherent way that supports the development of sustainable agriculture and ensures maximum impact for improving air quality, biodiversity, animal welfare and action on climate change.

Efforts to re-orientate resource use, consumption and waste patterns are needed to ensure societal-driven approaches to emission reductions. This includes better utilisation and understanding of the potential benefits of integrated livestock and crop systems. It also involves promoting the consumption of seasonal, and local and regional produce to reduce intensively reared livestock and cropping, promote the conservation of resources and prevent food waste. The EU should move towards a more coherent approach through the development of an EU-wide sustainable consumption and circular economy policy.

7. REFERENCES

- Amon, B., Kryvoruchko, V., Amon, T., et al., 2006. Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agric. Ecosyst. Environ. 112, 153–162.
- Amon, B., Kryvoruchko, V., Fröhlich, M., et al., 2007. Ammonia and greenhouse gas emissions from a straw flow system for fattening pigs: Housing and manure storage. Livest. Sci. 112, 199–207.
- Andersen, H.E., Blicher-Mathiesen, G., Bechmann, M., et al., 2014. Reprint of "Mitigating diffuse nitrogen losses in the Nordic-Baltic countries." Agric. Ecosyst. Environ. 198, 127–134.
- Aneja, V.P., Schlesinger, W.H., Erisman, J.W., 2009. Effects of Agriculture upon the Air Quality and Climate: Research, Policy, and Regulations. Environ. Sci. Technol. 43, 4234–4240.
- Bellarby, J., Tirado, R., Leip, A., et al., 2013. Livestock greenhouse gas emissions and mitigation potential in Europe. Glob. Chang. Biol. 19, 3–18.
- Burton, C.H., 2007. The potential contribution of separation technologies to the management of livestock manure. Livest. Sci. 112, 208–216.
- Bussink, D.W., Oenema, O., 1998. Ammonia volatilization from dairy farming systems in temperate areas: a review. Nutr. Cycl. Agroecosystems 51, 19–33.
- Butterbach-Bahl, K., Nemitz, E., Zaehle, S., et al., 2011. Chapter 19: Nitrogen as a threat to the European greenhouse balance, The European Nitrogen Assessment.
- Cederberg, C., 2004. Life Cycle Inventory of 23 Dairy Farms in South-Western Sweden. SIK Rapp. 728, 63.
- Cederberg, C., Hedenus, F., Wirsenius, S., et al., 2013. Trends in greenhouse gas emissions from consumption and production of animal food products - implications for long-term climate targets. Animal 7, 330–340.
- Chadwick, D., Sommer, S., Thorman, R., et al., 2011. Manure management: Implications for greenhouse gas emissions. Anim. Feed Sci. Technol. 166, 514–531.
- De Vries, M., de Boer, I.J.M., 2010. Comparing environmental impacts for livestock products: A review of life cycle assessments. Livest. Sci. 128, 1–11.
- Dentener, F., Stevenson, D., Ellingsen, K., et al., 2006. The Global Atmospheric Environment for the Next Generation. Environ. Sci. Technol. 40, 3586–3594.
- Diacono, M., Montemurro, F., 2010. Long-term effects of organic amendments on soil fertility. A review. Agron. Sustain. Dev. 30, 401–422.
- Edenhofer, O., Pichs-Madruga, R., Sokona, Y., et al., 2014. IPCC, 2014: Summary for Policymakers. Clim. Chang. 2014, Mitig. Clim. Chang. Contrib. Work. Gr. III to Fifth Assess. Rep. Intergov. Panel Clim. Chang. 1–40.
- EEA, 2014. Air quality in Europe – 2014 report.
- Erisman, J.W., Bleeker, A., Galloway, J., et al., 2007. Reduced nitrogen in ecology and the environment. Environ. Pollut. 150, 140–149.
- Erisman, J.W., Bleeker, A., Hensen, A., et al., 2008. Agricultural air quality in Europe and the future perspectives. Atmos. Environ. 42, 3209–3217.
- Fiala, N., 2008. Meeting the demand: An estimation of potential future greenhouse gas emissions from meat production. Ecol. Econ. 67, 412–419.
- Furger, M., Kunz, P., Schaffner, M., et al., 2013. Hochleistungskühe füttern: mit oder ohne Kraftfutter. Eth-schriftenr. zur Tierernährung 36, 11–25.
- Gattinger, A., Muller, A., Haeni, M., et al., 2012. Enhanced top soil carbon stocks under organic farming. Proc. Natl. Acad. Sci. 109, 18226–18231.
- Gattinger, A., Muller, A., Haeni, M., et al., 2013. Reply to Leifeld et al.: Enhanced top soil carbon stocks under organic farming is not equated with climate change mitigation. Proc. Natl. Acad. Sci. 110, E985.
- Gilhespy, S.L., Webb, J., Chadwick, D.R., et al., 2009. Will additional straw bedding in buildings housing cattle and pigs reduce ammonia emissions? Biosyst. Eng. 102, 180–189.
- Halberg, N., Hermansen, J.E., Kristensen, I.S., et al., 2010. Impact of organic pig production systems on CO_2 emission, C sequestration and nitrate pollution. Agron. Sustain. Dev. 30, 721–731.
- Hermansen, J.E., 2003. Organic livestock production systems and appropriate development in relation to public expectations. Livest. Prod. Sci. 80, 3–15.
- Hermansen, J.E., Kristensen, T., 2011. Management options to reduce the carbon footprint of livestock products. Anim. Front. 1, 33–39.
- Hocquette, J.-F., Chatellier, V., 2011. Prospects for the European beef sector over the next 30 years. Anim. Front. 1, 20–28.
- Hörtenhuber, S., Lindenthal, T., Amon, B., et al., 2010. Greenhouse gas emissions from selected Austrian dairy production systems – model calculations considering the effects of land use change. Renew. Agric. Food Syst. 25, 316–329.
- Hristov, A.N., Ott, T., Tricarico, J., et al., 2013. SPECIAL TROPICS – Mitigation of methane and nitrous oxide emissions from animal operations III: A review of animal management mitigation options. J. Anim. Sci. 91, 5095–5113.
- Hubbard, R., 2004. Water quality and the grazing animal. J. Anim. Sci. 82, 255–263.
- Janzen, H.H., 2011. What place for livestock on a re-greening earth? Anim. Feed Sci. Technol. 166-67, 783–796.

- Jongebreur, A., Monteny, G., 2001. Prevention and control of losses of gaseous nitrogen compounds in livestock operations: a review. ScientificWorldJournal. 1 Suppl 2, 844–51.
- Klevenhusen, F., Kreuzer, M., Soliva, C.R., 2011. Enteric and manure-derived methane and nitrogen emissions as well as metabolic energy losses in cows fed balanced diets based on maize, barley or grass hay. Animal 5, 450–61.
- Kristensen, T., Mogensen, L., Knudsen, M.T., et al., 2011. Effect of production system and farming strategy on greenhouse gas emissions from commercial dairy farms in a life cycle approach. Livest. Sci. 140, 136–148.
- Küstermann, B., Christen, O., Hülsbergen, K.-J., 2010. Modelling nitrogen cycles of farming systems as basis of site- and farm-specific nitrogen management. Agric. Ecosyst. Environ. 135, 70–80.
- Le, P.D., Aarnink, a J. a, Jongbloed, a W., et al., 2007. Effects of dietary crude protein level on odour from pig manure. Animal 1, 734–44.
- Leiber, F., 2014. Resigning protein concentrates in dairy cattle nutrition: a problem or a chance? Org. Agric. 4, 269–273.
- Leiber, F., Dorn, K., Probst, J.K., et al., 2015. Milk yields and protein efficiency of dairy cows at restricted concentrate feeding. Proc. Soc. Nutr. Physiol. Accept. Conf. Abstr.
- Leithold, G., Hülsbergen, K.-J., Brock, C., 2014. Organic matter returns to soils must be higher under organic compared to conventional farming. J. Plant Nutr. Soil Sci. In press.
- Lesschen, J.J.P., van den Berg, M., Westhoek, H.J., et al., 2011. Greenhouse gas emission profiles of European livestock sectors. Anim. Feed Sci. Technol. 166-167, 16–28.
- Meier, M., Böhler, D., Hörtenhuber, S., et al., 2014. Nachhaltigkeitsbeurteilung von Schweizer Rindfleischproduktionssystemen verschiedener Intensität. FiBL Reseach Inst. Org. Agric. 41, 82.
- Meier, M.S., Stoessel, F., Jungbluth, N., et al., 2015. Environmental impacts of organic and conventional agricultural products – Are the differences captured by life cycle assessment? J. Environ. Manage. 149, 193–208.
- Monteny, G.J., Bannink, A., Chadwick, D., 2006. Greenhouse gas abatement strategies for animal husbandry. Agric. Ecosyst. Environ. 112, 163–170.
- Montes, F., Meinen, R., Dell, C., et al., 2013. SPECIAL TOPICS – Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options. J. Anim. Sci. 91, 5070–5094.
- Nevison, C., 1998. Indirect N_2O Emissions from Agriculture, in: Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories. pp. 381–397.
- Nguyen, T.L.T., Hermansen, J.E., Mogensen, L., 2010. Environmental consequences of different beef production systems in the EU. J. Clean. Prod. 18, 756–766.
- Nousiainen, J., Shingfield, K.J., Huhtanen, P., 2004. Evaluation of milk urea nitrogen as a diagnostic of protein feeding. J. Dairy Sci. 87, 386–98.
- Novak, S., Fiorelli, J., 2010. Greenhouse gases and ammonia emissions from organic mixed crop-dairy systems: a critical review of mitigation options. Agron. Sustain. … 30, 215–236.
- O'Mara, F.P., 2012. The role of grasslands in food security and climate change. Ann. Bot. 110, 1263–70.
- Oenema, O., Wrage, N., Velthof, G.L., et al., 2005. Trends in global nitrous oxide emissions from animal production systems. Nutr. Cycl. Agroecosystems 72, 51–65.
- Pachauri, R.K., Allen, M.R., Barros, V.R., et al., 2014. Climate Change 2014: Synthesis Report. IPCC 138.
- Pachauri, R.K., Reisinger, A., Bernstein, L., et al., 2008. IPCC, 2007: Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Geneva, Switz. 104.
- Pardo, G., Moral, R., Aguilera, E., et al., 2014. Gaseous emissions from management of solid waste: a systematic review. Glob. Chang. Biol.
- Paulsen, H.M., Blank, B., Schaub, D., et al., 2013. Zusammensetzung , Lagerung und Ausbringung von Wirtschaftsdüngern ökologischer und konventioneller Milchviehbetriebe in Deutschland und die Bedeutung für die Treibhausgasemissionen 2013, 29–36.
- Philippe, F.-X., Laitat, M., Canart, B., et al., 2007. Gaseous emissions during the fattening of pigs kept either on fully slatted floors or on straw flow. animal 1, 1515–1523.
- Philippe, F.X., Laitat, M., Wavreille, J., et al., 2013. Influence of permanent use of feeding stalls as living area on ammonia and greenhouse gas emissions for group-housed gestating sows kept on straw deep-litter. Livest. Sci. 155, 397–406.
- Philippe, F.X., Nicks, B., 2013. Emissions of ammonia, nitrous oxide and methane from pig houses: Influencing factors and mitigation techniques. Agris 1–10.
- Powell, J.M., Misselbrook, T.H., Casler, M.D., 2008. Season and bedding impacts on ammonia emissions from tie-stall dairy barns. J. Environ. Qual. 37, 7–15.
- Reid, R.S., Thornton, P.K., Crabb, G.J.M.C., et al., 2004. Is it possible to mitigate Greenhouse Gas Emissions in pastoral Ecosystems of the Tropics? Environ. Dev. Sustain. 6, 91–109.
- Rockström, J., Steffen, W.L., Noone, K., 2009. Planetary boundaries: exploring the safe operating space for humanity. Ecol. Soc. 14, 1–32.
- Røjen, B. a, Lund, P., Kristensen, N.B., 2008. Urea and short-chain fatty acids metabolism in Holstein cows fed a low-nitrogen grass-based diet. Animal 2, 500–13.
- Schader, C., Jud, K., Meier, M.S., et al., 2014. Quantification of the effectiveness of greenhouse gas mitigation measures in Swiss organic milk production using a life cycle assessment approach. J. Clean. Prod. 73, 227–235.

- Schauberger, G., Lim, T.-T., Ni, J.-Q., et al., 2013. Empirical model of odor emission from deep-pit swine finishing barns to derive a standardized odor emission factor. Atmos. Environ. 66, 84–90.
- Schils, R.L.M., Eriksen, J., Ledgard, S.F., et al., 2013. Strategies to mitigate nitrous oxide emissions from herbivore production systems. Animal 7, 29–40.
- Sehy, U., Ruser, R., Munch, J.C., 2003. Nitrous oxide fluxes from maize fields: relationship to yield, site-specific fertilization, and soil conditions. Agric. Ecosyst. Environ. 99, 97–111.
- Smith, P., Bustamante, M., Ahammad, H., et al., 2014. Agriculture, Forestry and Other Land Use (AFOLU), in: Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. p. 179.
- Smith, P., Martino, D., Cai, Z., et al., 2008. Greenhouse gas mitigation in agriculture. Philos. Trans. R. Soc. Lond. B. Biol. Sci. 363, 789–813.
- Spek, J.W., Dijkstra, J., van Duinkerken, G., et al., 2013. Prediction of urinary nitrogen and urinary urea nitrogen excretion by lactating dairy cattle in northwestern Europe and North America: a meta-analysis. J. Dairy Sci. 96, 4310–22.
- Stavi, I., Lal, R., 2013. Agriculture and greenhouse gases, a common tragedy. A review. Agron. Sustain. Dev. 33, 275–289.
- Sutton, M., Howard, C., Erisman, J., 2011. The European nitrogen assessment: sources, effects and policy perspectives, Cambridge University Press.
- Taube, F., Gierus, M., Hermann, a., et al., 2014. Grassland and globalization - challenges for north-west European grass and forage research. Grass Forage Sci. 69, 2–16.
- Thomassen Van Calker, K.J., Smits, M.C.J., Iepema, G.L., De Boer, I.J.M., M.A., Thomassen, M. a., van Calker, K.J., et al., 2008. Life cycle assessment of conventional and organic milk production in the Netherlands. Agric. Syst. 96, 95–107.
- United-Nations, 2014. Guidance document on preventing and abating ammonia emissions from agricultural sources. ECE/EB.AIR/120.
- Van der Werf, H.M.G., Petit, J., Sanders, J., 2005. The environmental impacts of the production of concentrated feed: the case of pig feed in Bretagne. Agric. Syst. 83, 153–177.
- Velthof, G.L., Lesschen, J.P., Webb, J., et al., 2014. The impact of the Nitrates Directive on nitrogen emissions from agriculture in the EU-27 during 2000-2008. Sci. Total Environ. 468, 1225–1233.
- Webb, J., Broomfield, M., Jones, S., et al., 2014. Ammonia and odour emissions from UK pig farms and nitrogen leaching from outdoor pig production. A review. Sci. Total Environ. 470, 865–875.
- Wilkinson, J.M., 2011. Re-defining efficiency of feed use by livestock. Animal 5, 1014–22.
- Wuebbles, D., Hayhoe, K., 2002. Atmospheric methane and global change. Earth-Science Rev. 57, 177–210.
- Zehetmeier, M., Baudracco, J., Hoffmann, H., et al., 2012. Does increasing milk yield per cow reduce greenhouse gas emissions? A system approach. Animal 6, 154–66.
- Zeitz, J.O., Soliva, C.R., Kreuzer, M., 2012. Swiss diet types for cattle: how accurately are they reflected by the Intergovernmental Panel on Climate Change default values? J. Integr. Environ. Sci. 9, 199–216.
- Zhang, G., Strøm, J.S., Li, B., et al., 2005. Emission of Ammonia and Other Contaminant Gases from Naturally Ventilated Dairy Cattle Buildings. Biosyst. Eng. 92, 355–364.

Polyommatus icarus. *Source: Lukas Pfiffner, FiBL*

This publication is co-financed by the European Community, Directorate-General for the Environment. The sole responsibility for the publication lies with IFOAM EU. The European Commission is not responsible for any use that may be made of the information provided.