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DOTTORATO DI RICERCA IN PRODUZIONI ANIMALI

Sistemi produttivi e tecnologie di allevamento

LIFE CYCLE ASSESSMENT OF BOVINE MILK PRODUCTION IN NORTHERN ITALY

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WORK 2: **ENVIRONMENTAL IMPACT OF DIFFERENT DAIRY FARMING SYSTEMS IN DENMARK, GERMANY AND ITALY.**

WORK 3: **CARBON FOOTPRINT OF MILK PRODUCTION IN ITALIAN ALPS THROUGH A SENSITIV APPROACH.**

WORK 4: **HOW DIFFERENT LEVELS OF INTENSIFICATION CAN AFFECT THE ENVIRONMENTAL IMPACT OF MILK PRODUCTION? A STUDY ON ITALIAN FARMS.**

1. INTRODUCTION

1.1. AN OVERVIEW OF WORLD LIVESTOCK SITUATION AND TREND

Growing population and other demographic factors such as age structure and urbanization determine food demand and have driven the intensification for agriculture for centuries. Growing economy and individuals income have also contributed to growing demand and a shift in diets (Steinfeld et al., 2006). These increase in demand is particularly strong in developing countries, between the 1960s and 2005 (Figure 1) for example, annual per capita consumption of meat more than tripled, that of milk almost doubled, while per capita consumption of eggs increased fivefold in the developing world (FAO, 2009).

Between 1980 and 2007, global production of meat, milk and eggs has increased at annual rate of 3%, 1.4% and 3.4%, respectively in the period 1980–2007. Growth in production has been fastest in developing countries where production of meat, milk and eggs has been growing at 5% and 4% and 6.3% p.a. during the period 1980–2007. In comparison, meat, milk and egg output in developed countries has been growing at only 0.8%, 0.1% and 0.2% p.a. over the same period (Opio et al., 2012).

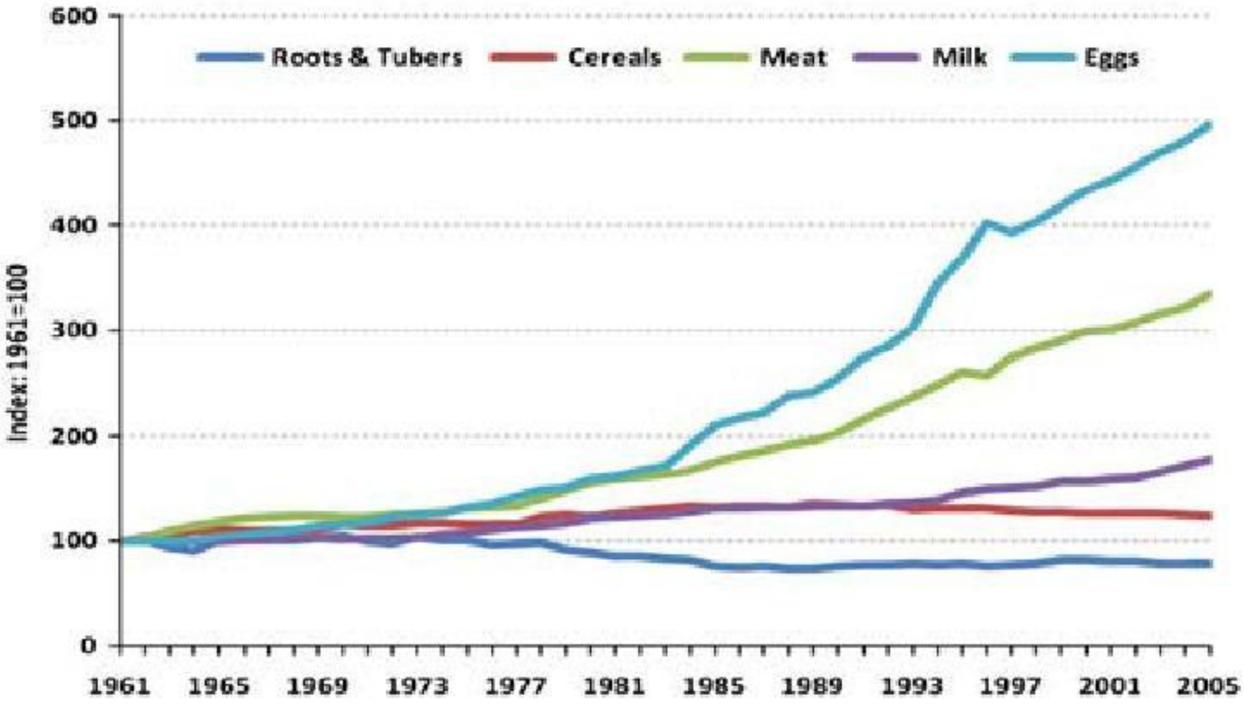


Fig. 1 - Per capita consumption of major food items in developing countries (Opio et al., 2012).

The increase of demand of animal products driven by growing population and incomes is stronger than for most food items. Global production of meat is projected to more than double from 229 million tonnes 1999/2001 to 465 million tonnes in 2050, and that of milk to increase from 580 to 1043 million tonnes (Gerber and Steinfeld, 2010).

On average, in 2050, each person on the planet will be consuming 52 kg of meat and 115 kg of milk a year, considerably more than consumption levels today, Figure 2 and 3 show that by 2050, developing world peoples are projected to consume 44 kg of meat and 78 kg of milk annually (Garnett, 2009).

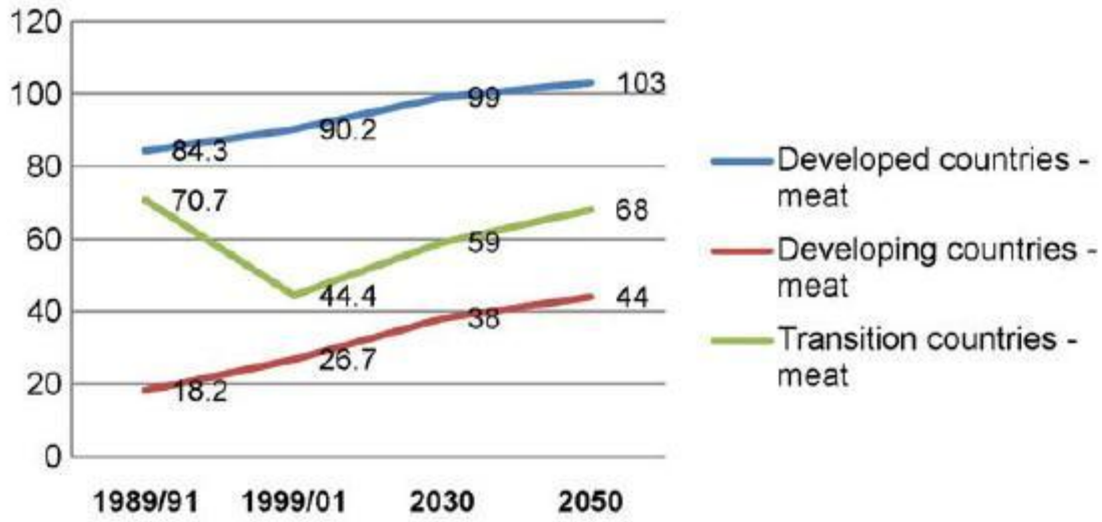


Fig. 2 - Projected trends in per capita consumption of meat products to 2050 kg/person/yr. Source: FAO (2006) World Agriculture: Towards 2030/2050.

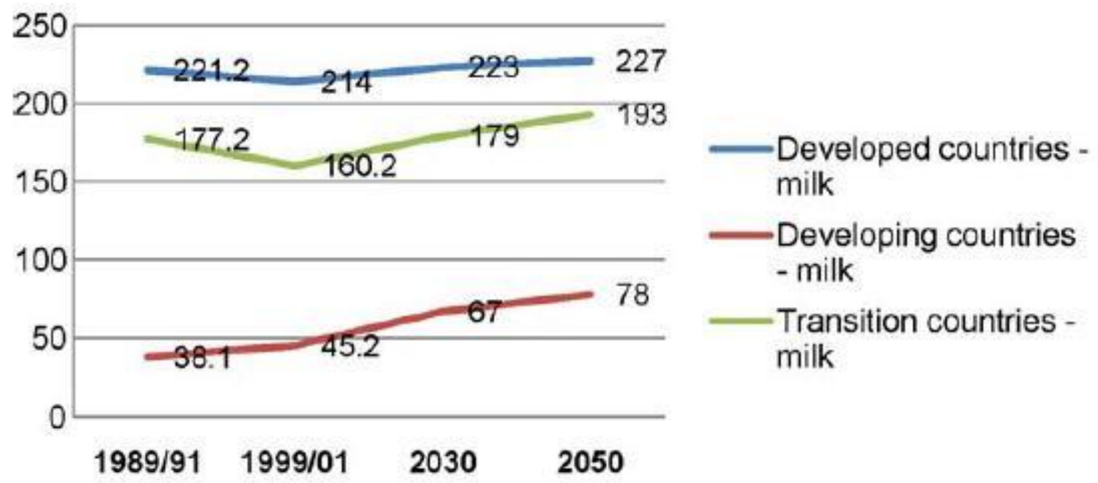


Fig. 3 - Projected trends in per capita consumption of milk products to 2050 kg/person/yr. Source: FAO (2006) World Agriculture: Towards 2030/2050.

The agriculture sector has responded to the increased and diversified demands for foods items with innovation in biology, chemistry and machinery. It has done so mainly through intensification rather than expansion. Land use has changed correspondingly (Steinfeld et al., 2006).

In animal production, technological development has been most rapid in those subsectors that have experienced the fastest growth: broiler and egg production, pork and dairy. However, certain key technological changes have occurred in the production of all livestock commodities - a grow production intensity, characterized by increasing use of feed cereals, use of advanced genetics and feeding systems, animal health protection and enclosure of animals (Steinfeld et al., 2006). The growth and industrialization of the livestock sector would not have been possible without a concurrent increase in crop production. As evidenced from the trends in consumption and production, the overall trend in the livestock sector has been a transition from land-based extensive ruminant systems to large-scale industrial non-ruminant systems that rely on concentrate feed. The shift towards industrialized mono-gastric production explains the increased use of concentrate feed by the sector (Opio et al., 2012). Figure 4 provides an overview of the use of concentrate feed in 1970 and 2007.

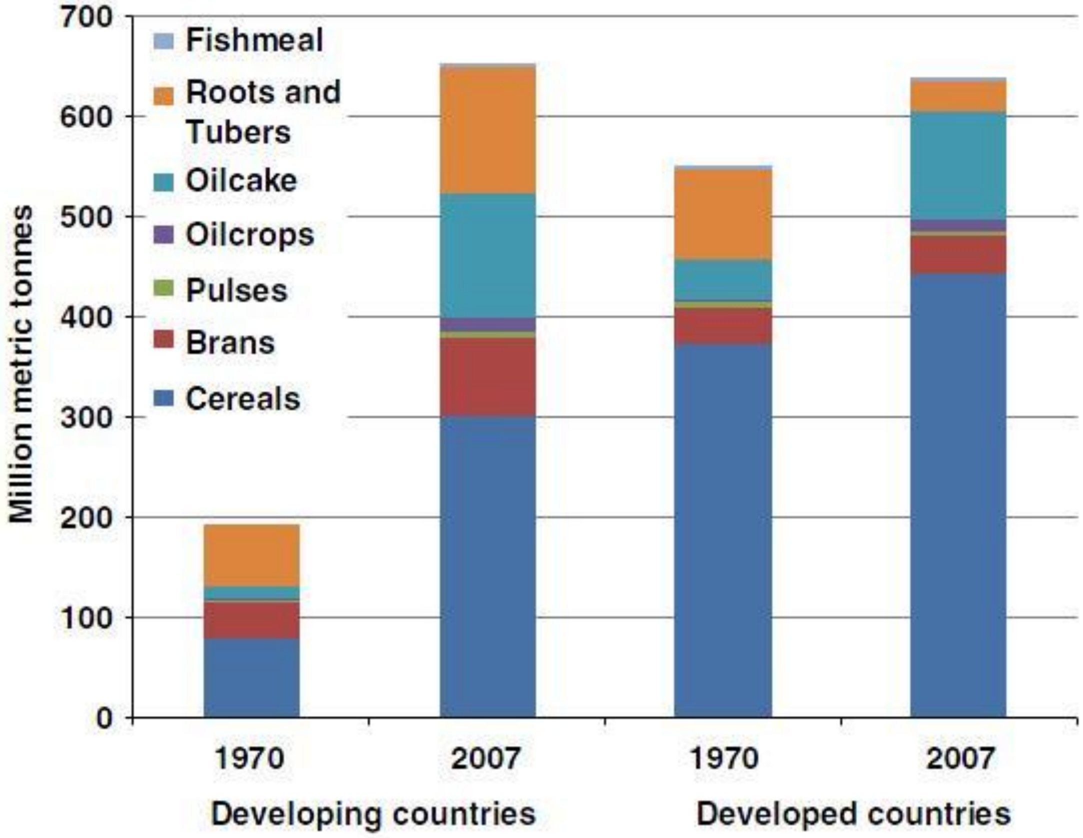


Fig. 4 - Trends in concentrate feed use in developing and developed countries (Opio et al., 2012).

The main input to livestock rearing is land. Directly and indirectly, through grazing and through feed-crop production, the livestock sector is a major user of natural resources such as land and water, using about 35% of total land and about 20% of green water for feed production (Deutsch et al., 2010).

Additional land available for cultivation is limited. Therefore, most of the increase in agricultural production has come, and will come, from intensification of land that is already cropped or grazed. As a large user of crops and other plant material, the livestock sector must continue to improve the conversion of these materials into edible products. In 2002, a total of 670 million tonnes of cereals were fed to livestock, representing roughly one third of the global cereal harvested. Another 350 million of tonnes protein-rich processing by-products are used as feed (mainly brans, oilcakes and fishmeal). The use of feed concentrates for ruminants is limited to countries where meat prices are high relative to grain prices. Intensification accounts for the bulk of supply expansion over the past 25 years, and it is result of technological advanced and higher input use in crop production - notably plant breeding, the application of fertilizers and mechanization (FAO, 2012).

The livestock sector has a primary and growing role in the agricultural economy. It is a major provider of livelihoods for the larger part of the world's poor. It is also an important determinant of human health and component of diets. But already the livestock sector is a source of instability to many ecosystems and contributes to global environmental problems. The future of the livestock environment interface will be shaped by how we resolve the balance of two competing demands: one for animal food products and the other for environmental service. Both demand are driven by the same factors: increasing population, growing income and urbanization (Gerber and Steinfeld, 2010). Livestock production has direct and indirect impacts on the environment, ranging from land degradation due to overgrazing, through to biodiversity loss mainly brought about by land use and land-use change and ecosystem pollution, to air, water and land pollution from animal waste and noxious emissions and climate change (Opio et al., 2012).

1.2. ENVIRONMENTAL IMPACT OF LIVESTOCK PRODUCTION

1.2.1. Global warming potential of the livestock sector

Climate change means an increase in average temperature and seems to be associated with an increased frequency of extreme weather events. Average temperature have increased by 0.8°C over the past century (Steinfeld et al., 2006).

The greenhouse effect is a key mechanism of temperature regulation. Without it, the average temperature of the earth's surface would not be 15°C but -6°C. The earth returns energy received from the sun back to space by the reflection of light and by emission of heat. A part of the heat flows is absorbed by so-called greenhouse gases (GHG), trapping it in the atmosphere. The principal greenhouse gases involved in these process include carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) and chlorofluorocarbons (Steinfeld et al., 2006). These gases are substances for which the Intergovernmental Panel on Climate Change (IPCC) has defined a global warming potential coefficient. They are expressed in mass-based CO_2 equivalents (CO_2 eq.). Since the GWP factors have changed during the years, the most current IPCC GWP factors can be found from the "The Physical Science Basis" section of the IPCC 2007 (Forster et al., 2007) report from "Technical Summary" chapter. The most common characterization factors for GWP in 100 years-time horizons are:

1 kg of carbon dioxide (CO_2) = 1 kg CO_2 eq.

1 kg of methane (CH_4) = 25 kg CO_2 eq.

1 kg of nitrous oxide (N_2O) = 298 kg CO_2 eq.

CH_4 has an atmospheric lifetime of 12 years and has therewith a much higher contribution to global warming potential in a short term, but after 20-30 years this effect decays almost completely. Compared to a 100 year perspective, the characterization factor for CH_4 is three time higher in 20 year perspective (72 kg CO_2 eq.) and three times lower in a 500 year (7.6 kg CO_2 eq.). For N_2O the difference in characterization factor is small in a 20 year perspective (298 kg CO_2 eq.) but half in 500 year perspective (153 kg CO_2 eq.), compared to a 100 year perspective (Flysjö, 2012).

Livestock activities also emit considerable amounts of these three gases. In 2006, the Food and Agriculture Organization published "Livestock's Long Shadow" (Steinfeld et al., 2006) Environmental Issues and Options, which provided the first-ever global estimates of the livestock sector's contribution to GHG emissions. Taking into account the entire livestock food chain, the study estimated this contribution to be about 18% of total anthropogenic emissions. For the agriculture sector alone, livestock constitute nearly 80% of all emission. Livestock contribute about 9% of total carbon dioxide (CO_2) emissions, but 37% of methane (CH_4) and 65% of nitrous oxide (N_2O).

The livestock products are GHG intensive compared with other food groups, and that the vast majority of impacts occur at the farm stage, with subsequent processing, retailing and transport playing more minor roles (Garnett, 2009).

More recent and disaggregated FAO estimates on the sector's contribution to global anthropogenic GHG emissions highlights the differences between species with beef production contributing about 5.5% of total global anthropogenic emissions while milk and pork contribute 2.8% and 1.9%, respectively (Figure 5 and 6) (Opio et al., 2012).

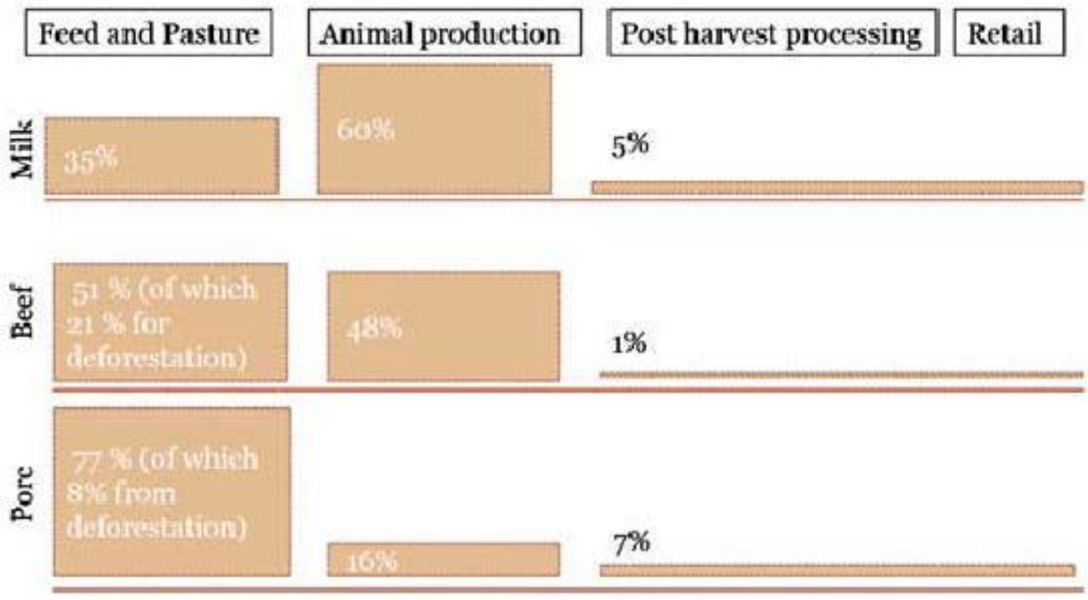


Fig. 5 - Greenhouse gas emissions along livestock food chains (Opio et al., 2012).

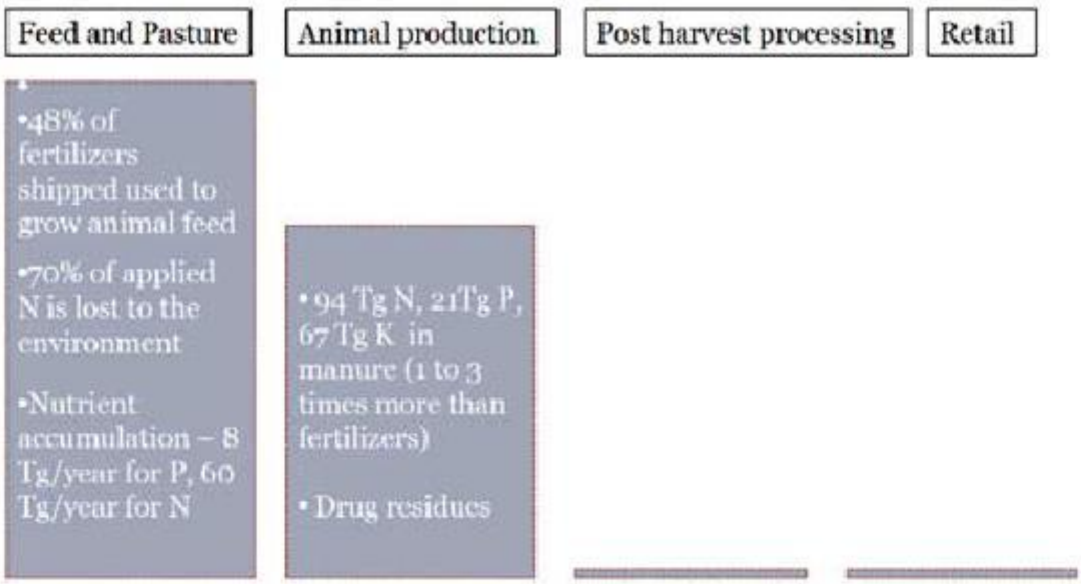


Fig. 6 - Impact on nutrient flows along livestock food chains (Opio et al., 2012).

On a global scale, the emission intensity of meat and milk, measured by output weight, corresponds on average to 60 kg CO_2 eq. kg^{-1} of CW and 9.4 kg CO_2 eq. kg^{-1} of carcass weight (CW), for beef and pork, respectively, and 2.4 kg CO_2 eq. kg^{-1} of milk (Gerber et al., 2010). This analysis also found significant variability in emissions across the different world regions (Opio et al., 2012).
Direct emissions from livestock come from the respiratory process of all animal in the form of carbon dioxide, but these makes up only very small part of the net release of carbon. That amount is considered a part of a rapidly cycling biological system where the plant matter consumed was itself created through the conversion of atmospheric CO_2 in organic compounds. Since the emitted

and the absorbed quantities are considered to be equivalent, livestock respiration is not considered to be a net source under the Kyoto Protocol (Steinfeld et al., 2006).

1.2.1.1. Methane

Globally CH_4 emissions make up around 14% of the GHG emission induced by human activities (Barker et al., 2007).

Ruminants, emit methane as a part of their digestive process, which involves microbial fermentation of fibrous feed. In the rumen, or large-fore stomach, of these animals, microbial fermentation converts fibrous feed into products that can be digested and utilized by animals. These microbial fermentation process, referred to as enteric fermentation, produce methane as by product, which is exhaled by the animal. Methane emission are determined by production systems and regional characteristics. They are affected by energy intake and several other animal and diet factors (quantity and quality of feed, animal body weight, age and amount of exercise). Animal manure also emits gases such as methane, depending on the way they are produced (solid, liquid) and managed (collection, storage, spreading). This occurs mostly when manure is managed in liquid form, such as in lagoons or holding tanks. Globally, livestock are the most important source of anthropogenic methane emissions. Together, enteric fermentation and manure represent some 80% of agricultural methane emission and about 35-40% of total anthropogenic methane emissions (Steinfeld et al., 2006).

1.2.1.2. Nitrous oxide

N_2O emissions may be related to the use of both organic and inorganic fertilizers, biological nitrogen fixation, and return of crop residues to the field or to animal production (EEA, 2011b).

Generally only a small portion of the total nitrogen excreted is converted to N_2O during handling and storage of managed waste (Steinfeld et al., 2006). For the N_2O emission to occur, the waste must first be handled aerobically, allowing ammonia or organic nitrogen to converted to nitrates and nitrites (nitrification). The amount of N_2O released during storage and treatment of animal wastes depends on the system and duration of waste management and the temperature. There is an antagonism between emission risks of methane versus nitrous oxide for the different waste storage pathways - trying to reduce methane emissions may well increase those of N_2O.

The application of nitrogen containing fertilizers is the major component leading to N_2O emissions from soils. Beside the amount of applied nitrogen, the amount of N_2O emitted is also influenced by nitrogen application and irrigation practices, climatic variables, soil temperature and humidity. Additionally, cultivation techniques also lead to N_2O emissions from soil due to the mineralization of organic matter (EEA, 2011b).

Manure-included soil emission are clearly the largest livestock source of N_2O worldwide. Emission fluxes from animal grazing (unmanaged waste, direct emission) and from the use of animal waste as fertilizers on cropland are of a comparable magnitude (Steinfeld et al., 2006).

What share of direct emissions from fertilizer can we attribute to livestock? A large share of the world's crop production is fed to animals and mineral fertilizers is applied to much of the corresponding cropland. Intensively managed grassland also receive a significant portion of mineral fertilizer. The FAO/IFA (2001) calculations result in a mineral fertilizer N_2O-N loss rate of 1%.

N_2O emissions result also from the re-deposited fraction of volatilized NH_3 from manure and the application of mineral fertilizer that reached the aquatic reservoirs and from the fraction of nitrogen leached in the soil.

Livestock activities contribute substantially to the emission of nitrous oxide, the most potent of the three major greenhouse gases. They contribute almost two-thirds (65%) of all anthropogenic N_2O emission and 75-80% of agricultural emission (Steinfeld et al., 2006).

1.2.1.3. Carbon dioxide from land use change

Some 11.7% of the global land area (that is not covered with ice) is in agricultural use at the moment. It has been proposed that this should not exceed 15% because it is estimated that the agricultural activities would need to expand to less productive areas if this limit is exceeded. This would lead to intensification of deforestation which would have an adverse impact on the essential ecosystem services (Rockström et al., 2009).

Some countries have seen a particularly strong expansion of area cropped, most of it at the expense of forest (Brazil and other Latin American countries). Much of these area expansion has been for the production of concentrate feeds for livestock, notably soybeans and maize. Through the expansion of land for livestock development, sector growth has been a prime force in deforestation in Latin America and the Caribbean, and in overgrazing in other regions (FAO, 2012).

Land use change or land transformation means, for example, the change from forestry to agriculture, but also from one agricultural purpose, e.g. from meadow to field. Land use causes various environmental impacts. At the moment the focus is on land use related greenhouse gas emissions, but changes in carbon cycles and storages, soil quality and soil net productivity, and loss of biodiversity are growing in importance. Additionally, changes in land use and land cover also affect water quality and availability. The IPCC has estimated that the land use change is the second most important source of GHG emissions, right after the use of fossil fuels (Mattila et al., 2011).

Change in land use have an impact on carbon fluxes and many of the land use changes involve livestock, either occupying land (as pasture or arable land for feed-crops) or releasing land for other purposes, when, for example, marginal pasture land is converted to forest. A forest contains more carbon than does a field on annual crops or pasture, and so when forests are harvested, or worse, burned, large amounts of carbon are released from the vegetation and soil to the atmosphere. Livestock's role in deforestation is of proven importance in Latin America, the continent suffering the largest net loss of forests and resulting carbon fluxes. globally, livestock induced land use change generates 2.4 billion tonnes of CO_2 a year, equivalent to approximately 7% of global GHG emissions (Steinfeld et al., 2006). These emissions arise not only from soy production but also from the cultivation of other feed crops, and from the encroachment of grazing into forested areas (Garnett, 2009). Soy cultivation is a major driver of deforestation in the Brazilian Amazonian region (Nepstad et al., 2006; WWF, 2004). In the decade up to 2004, industrial soybean farming doubled its area to 22,000 km^2 and is now the largest arable land user in Brazil (Elferink et al., 2007). It has been estimated that the annual net emissions from Brazilian Amazonian deforestation, based on the average deforestation rate of 19,400 km^2 per year for the 2007 period, was approximately 191 million tonnes of CO_2 eq. carbon, or 700 million tonnes CO_2 eq. (McAlpine et al., 2009). This represents more than 2% of global GHG emissions. The main cause is cattle ranching and in this case the link between land use change derived CO_2 release and livestock production is very direct (USDA, 2008). The EU represents a major export market, accounting for 32% of Brazil's soy animal feed exports in 2006/2007; producing this volume has been calculated to require 50,000 km^2 of Brazilian land (Garnett, 2009). In a recent report Gerber et al., (2010) estimated 7.69 kg CO_2-eq. per kg of soybean cake from soybeans produced in Brazil, entirely associated with deforestation, for which land use change emissions were accounted.

Land use and land se change (LULUC) can have also a positive impact (Flysjö, 2012). Enhancing carbon sequestration in soil is identified as the most promising mitigation strategy for agriculture

(Barker et al., 2007). Because of the extensive nature of grasslands, they hold enormous potential to serve as one of the greatest terrestrial sinks for carbon (Opio et al., 2012). However the sequestration potential is debatable (Flysjö, 2012) and some research shows there is a large potential for carbon storage (Soussana et al., 2007), moreover the uncertainty about the potential of several measures to store additional carbon (i.e. no tillage or conservation tillage, use of manure on crop instead of grassland, improved rotations with higher C input to soil (catch crop), increased crop yield and hence the related crop residues, for example, by better plant breeding, crop husbandry, irrigation or fertilization and conversion from arable land to grassland or grazing management), the rate of accumulation of soil carbon, and the permanence of this carbon sink is high. In addition, their life cycle budget regarding all GHGs is unknown (de Boer et al., 2011).
Clearly, estimating CO_2 emissions from land use and land-use change is far less straightforward than those related to fossil fuel combustion. It is even more difficult to attribute these emissions to a particular production sector such as livestock (Steinfeld et al., 2006).

1.2.1.4. Greenhouse gas emissions in the global dairy sector

During the half past century (1961-2009) global milk production has increased by 86%, both number of dairy cows and the milk yield per cow have increased (by 42 and 31%, respectively). The production growth has been lower in Europe and North America (by 10 and 44%, respectively) as the result of significantly higher milk yield per cow (130 and 186%, respectively) but from a reduced number of dairy cows. In Asia milk production increased by more 600% and in Latin America was more than quadrupled, these trend are the results of both a larger number of animals and higher milk yields. Oceania has more than doubled its milk production, primarily due to an increased milk yield per cow (74%), but also a slight increase in the number of animals (Flysjö, 2012).
The FAO report "Greenhouse Gas Emissions from the Dairy Sector" (Gerber et al., 2010) attributes to the milk and meat production from the dairy herd (comprising of milking cows, replacement calves and surplus calves and culled animals) plus the processing of dairy products, production of packaging and transport activities to contribute 4% to the total global anthropogenic GHG emissions (±26%).
If is taken into account only the emissions associated with milk production, processing and transportation of milk and milk products the overall to total anthropogenic emissions is estimated at 2.7% (±26%). The average global emissions from milk production, processing and transport is estimated to be 2.4 CO_2-eq. per kg of FPCM (fat protein corrected milk) at farm gate (±26%) (Table 1). Globally, cradle to farm gate emissions contribute, on average, 93% of total dairy GHG emissions. On-farm activities (including land use change) contribute most significantly to overall GHG emissions. In industrialized countries, the relative contribution ranges between 78 and 83% of total life cycle emissions, while in developing world regions the contribution is much higher – ranging between 90 and 99% of total emissions (Figure 7).
Methane contributes most to the global warming impact of milk, about 52% of the GHG emissions. Nitrous oxide emissions account for 27 and 38% of the GHG emissions in developed and developing countries, respectively, while CO_2 emissions account for a higher share of emissions in developed countries (21%), compared to developing countries (10%).
The grassland based and mixed systems are both estimated to contribute around 50% to global milk production. However, grassland based systems, on average, account for 60% of the global sector's emissions, whereas mixed systems are characterized by a lower emission intensity, and are thus estimated to account for only 40% of emissions. The average emissions from grassland based systems are 2.72 kg CO_2 eq. per kg of FPCM, compared to an average of 1.78 kg CO_2 eq. per kg of FPCM, in the mixed systems.

Tab. 1 – Milk and neat production and related GHG emissions – global averages (Gerber et al., 2010).

Commodities	Total production (Million tonnes)	GHG emissions (Million tonnes CO_2-eq.) *	GHG emissions (kg CO_2.eq. per kg of product) *	Contribution to total anthropogenic emissions in 2007 (%) *
Milk: production, processing and transport	553	1 328	2.4	2.7
Meat: produced from slaughtered dairy cows and bulls (carcass weight)	10	151	15.6	0.3
Meat: produced from fattened surplus calves (carcass weight)	24	490	20.2	1.0

* [±26 percent]

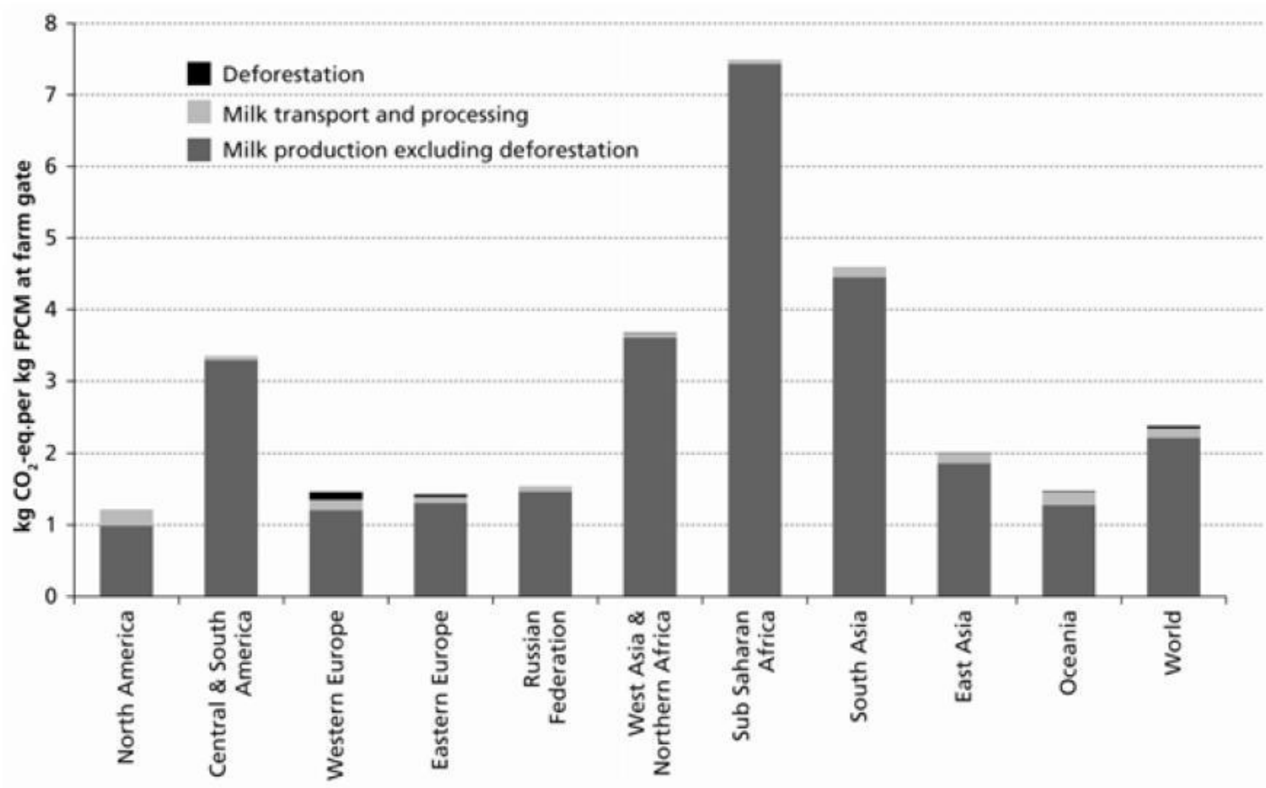


Fig. 7 - Estimated GHG emissions per kg of FPCM at farm gate, averaged by main regions and the world (Gerber et al., 2010).

1.2.2. The role of livestock in eutrophication and water pollution

High concentration of nutrients in water resources can lead to over-stimulation of aquatic plant and algae growth leading to eutrophication. Eutrophication is a natural process in the ageing of

lakes and some estuaries, but livestock and other agriculture activities can greatly accelerate eutrophication by increasing the rate of witch nutrients and organic substances enter aquatic ecosystems from their surrounding watersheds (Steinfeld et al., 2006).
In severe cases of eutrophication, massive blooms of algae (sessile and planktonic) occur. Some blooms are toxic. As dead algae decompose, the oxygen in the water is used up; bottom-dwelling animals die and fish either die or leave the affected area. In severe cases of eutrophication, massive blooms of algae (sessile and planktonic) occur. Some blooms are toxic. As dead algae decompose, the oxygen in the water is used up; bottom-dwelling animals die and fish either die or leave the affected area (EAA, 2000).
Nutrient intake by animals can be extremely high, for example a productive dairy cow ingests up to 163.7 kg of N and 22.6 kg of P per year and it excretes 129.6 kg of N and 16.7 kg of P (respectively the 79 and the 73% of each nutrient ingested) in form of manure that may represent a threat to water quality. The efficiency as the amount of N harvested from the world's cropland with respect of the annual N input, results in the even lower efficiency of some 40%. This result is affected by animal manure, which has a relatively high loss rate as compared to mineral fertilizer. Mineral fertilizer is more completely absorbed, depending of the fertilizer application rate and the type of mineral fertilizer. Most of N losses are not directly emitted to the atmosphere, but enter the N cascade through water (Steinfeld et al., 2006).
Leaching is a mechanism whereby N applied to the soil is lost to water resources. In its nitrate (NO_3) from (inorganic N), nitrogen is very mobile in soil solution, and can be easily be leached below the rooting zone to groundwater, or enter the subsurface flow. Nitrogen (especially its organic form) can also be carried into water system through run off.
High concentrations of nitrate in drinking water are considered a human-health problem because in the stomach nitrate is converted rapidly to nitrite, which can cause a reduction in the blood's oxygen-carrying capacity. The WHO guide value for nitrate concentration in drinking water is 45 mg l^{-1} (10 mg NO_3-N) (Steinfeld et al., 2006).
Phosphorus in water is not considered to be directly toxic to humans and animals and, therefore, no drinking water standards have been established for P. Phosphorus contaminates water resources when manure is directly deposited or discharged into the stream or when excessive levels of phosphorus are applied to the soil. Unlike nitrogen, phosphorus is held by soil particles and is less subject to leaching unless concentration levels are excessive. Erosion is in the fact the main source of phosphate loss and phosphorus is transported in surface run off in soluble or particulate form (Steinfeld et al., 2006).
The livestock sector is the major cause of these increase, in some countries (i.e. Canada, France, Germany, the United Kingdom and the United States) livestock are directly or indirectly responsible for more than 50% of the mineral N and P applied on agricultural land (Steinfeld et al., 2006).

1.2.3. The role of livestock in acidification

Over greenhouse gases, the livestock sector is an important source of other air pollutants as ammonia, nitrogen oxides, sulphur dioxide and volatile organic compounds. In the presence of atmospheric moisture and oxidants, sulphur dioxide and oxides of nitrogen are converted to sulphuric and nitric acids. These airborne are noxious to respiratory system and attacks some materials. These air pollutants return to earth in the form of acid rain and snow, and as a dry deposited gases and particles, which may damages crops and forests and makes lake and streams unsuitable for fish and other plant and animal life (Steinfeld et al., 2006).
As a secondary particulate precursor, NH_3 also contributes to the formation of particulate aerosols in the atmosphere. In particular NH_3 contributes to acid deposition and eutrophication which in

turn can lead to potential changes occurring in soil and water quality. In many cases, the deposition of acidifying and eutrophying substances still exceeds the critical loads of the ecosystems (EAA, 2011a).
Some 94% of global anthropogenic atmospheric emission of ammonia is produced by the agricultural sector. The livestock sector contributes about 68 percent of the agriculture share, mainly from deposited and applied manure (Steinfeld et al., 2006).
During storage (including the preceding excretion in animal houses) the organically bound nitrogen in feaces and urine starts to mineralize to NH_3/ NH_4^+, providing the substrate for nitrifiers and denitrifiers (and hence, eventual production of N_2O). For the most part these excreted N compounds mineralize rapidly. In urine, typically over 70% on the nitrogen is presented as urea (IPCC, 1997).
Turning to ammonia, rapid degradation to urea and uric acid to ammonium leads to very significant N losses through volatilization during storage and treatment manure. While actual emissions are subject to many factors, particularly the manure management system and ambient temperature, most of the NH_3-N volatilizes during storage (typically about one-third of the initially voided N) and before application or discharge (Steinfeld et al., 2006).
Excreta freshly deposited on land (either applied by mechanical spreading or direct deposition by livestock) have high nitrogen losses rate, resulting in substantial ammonia volatilization (Steinfeld et al., 2006). FAO/IFA (2001) estimate the N loss via NH_3 volatilization from animal manure, after application, to be 23% worldwide.
What share of direct emission from fertilizer can we attribute to livestock? A large share of the world's crop production is fed to animals and mineral fertilizer is applied to much of the corresponding cropland: 20 to 25% of mineral fertilizer use (about 20 million tonnes N) can be ascribed to feed production for the livestock sector. The average mineral fertilizer NH_3 volatilization loss rate is 14% (FAO/IFA, 2001). On these basis, livestock production can be considered responsible for a global NH_3 volatilization from mineral fertilizer of 3.1 million tonnes.

1.2.4. The role of livestock in water depletion

The Water Footprint (WF) is a measure of humans' appropriation of freshwater resources and has three components: blue, green, and gray. The blue WF refers to consumption of blue water resources (surface and ground water), whereby consumption refers to the volume of water that evaporates or is incorporated into a product. The blue WF is thus often smaller than the water withdrawal, because generally part of a water withdrawal returns to the ground or surface water. The green WF is the volume of green water (rainwater) consumed, which is particularly relevant in crop production. The gray WF is an indicator of the degree of freshwater pollution and is defined as the volume of freshwater that is required to assimilate the load of pollutants based on existing ambient water quality standards. Agricultural production takes a share, accounting for 92% of the global WF (Hoekestra et al., 2012).
Livestock use of water and contribution to water depletion trends are high growing. An increasing amount of water is needed to meet growing water requirements in livestock production process, from feed production to product supply (Steinfeld et al., 2006). FAO (2003) estimates that about 80% of projected growth in crop production in developing countries will come from intensification in form of yield increase (67%) and higher cropping intensities (12%). It is estimated that in the developing countries at present, irrigated agriculture, with about a fifth of all arable land, account for 40% of all crop production and almost 60% of cereal production. The area equipped for irrigation in developing countries is projected to expand by 40 million hectares (20%) over the projection period. These underlines the importance of the livestock sector's responsibility for irrigation water use.

The water footprint of a live animal consists of different components: the indirect water footprint of the feed and the direct water footprint related to the drinking water and service water consumed (Chapagain and Hoekstra, 2003). The water footprints of animal products vary greatly across countries and production systems but looking to a global average the water footprint of meat the water footprint of meat increases from chicken meat (4,300 m^3 t^{-1}), goat meat (5,500 m^3 t^{-1}), pig meat (6,000 m^3 t^{-1}) and sheep meat (10,400 m^3 t^{-1}) to beef (15,400 m^3 t^{-1}). The differences can be partly explained from the different feed conversion efficiencies of the animals (Mekonnen et al., 2012). For all farm animal products, except dairy products, the total water footprint per unit of product declines from the grazing to the mixed production system and then again to the industrial production system. The reason is that, when moving from grazing to industrial production systems, feed conversion efficiencies get better. Per unit of product, about three to four times more feed is required for grazing systems when compared to industrial systems animals (Mekonnen et al., 2012). That explains why, despite the larger WF of concentrate feed production compared to roughages, the water footprint in a product perspective is worse in the grazing systems. During the period 1996–2005, the total water footprint for global animal production was 2,422 Gm^3 y^{-1} (87.2% green, 6.2% blue and 6.6% grey water). The largest water footprint for animal production comes from the feed they consume, which accounts for 98% of the total water footprint. Drinking water, service water and feed-mixing water further account the only for 1.1, 0.8 and 0.03% of the total water footprint, respectively. Grazing accounts for the largest share (38%), followed by maize (17%) and fodder crops (8%) (Mekonnen et al., 2012).

1.2.5. The role of livestock in biodiversity losses

Biodiversity is a concept with a wide content, and in the Convention on Biological Diversity it is stated that *'Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems'* (UNEP 1992). The most important drivers of biodiversity loss and ecosystem service changes are habitat change, climate change, invasive alien species, overexploitation and pollution.

Habitat destruction, fragmentation and degradation are considered the major category of threat to global biodiversity.

Livestock are one of the major drivers of habitat change (deforestation, destruction of riparian forests, drainage of wetlands), due to livestock production itself or for feed production. Livestock also directly contribute to habitat change as overgrazing and overstocking accelerate desertification.

It is currently difficult to be precise when quantifying livestock-induced biodiversity loss. Losses are the result of a complex web of changes, occurring at different levels, each of which is affected by multiple agents (Steinfeld et al., 2006). In particular biodiversity has been negatively influenced by intensive agriculture, forestry and the increase of urban areas and infrastructure. The measurement of land use impacts on biodiversity, however, is a complex task, because a widely accepted definition of biodiversity does not exist (Koellner and Scholz, 2008).

1.3. THE EUROPEAN SITUATION AND TRENDS

1.3.1. Greenhouse gases emission in the European livestock sector.

In 2008, the 27 member states of the European Union (EU-27) produced 26%, 13%, 22%, 12% and 11% of the world’s milk, beef, pork, poultry and eggs, respectively. Following considerable growth in the 1960s and 1970s, cattle numbers in Europe have been decreasing since 1980s. The number

of pigs in the EU has stabilized since the middle 1980s, whereas the number of poultry is increasing (Lesschen et al., 2011). EU 27 GHG emissions from agriculture accounted for 10% of total GHG emissions in 2008, contrary to the energy related sectors, which are dominated by CO_2 emissions, N_2O (58%; mainly from plant production) and CH_4 (42%; mainly from animal husbandry) are the predominant GHGs in agriculture. Between 1990 and 2008, they decreased by 20% (EEA, 2011b).
Enteric fermentation of cattle is the largest source of CH_4 emissions in the EU27 and represented 31% of total GHG emissions from the agriculture sector in 2008 (EEA, 2011b) and in the EU-15 accounting for 2.8% of total GHG emissions in 2009 (EEA, 2011c).
Between 1990 and 2008, CH_4 emissions from enteric fermentation of cattle decreased continually due to a declining number of cattle (between 1990 and 2009, CH_4 emissions from enteric fermentation from cattle declined by 12% in the EU-15 (EEA, 2011c)). Although higher milk yields were achieved through increased energy intake per cattle, among other measures, which resulted in a higher amount of CH_4 emitted per cow (emission intensity), the decrease in the number of cattle within the EU had a larger effect on total CH_4 emissions and drove down emissions from this sector (EEA, 2011b) (Figure 8).

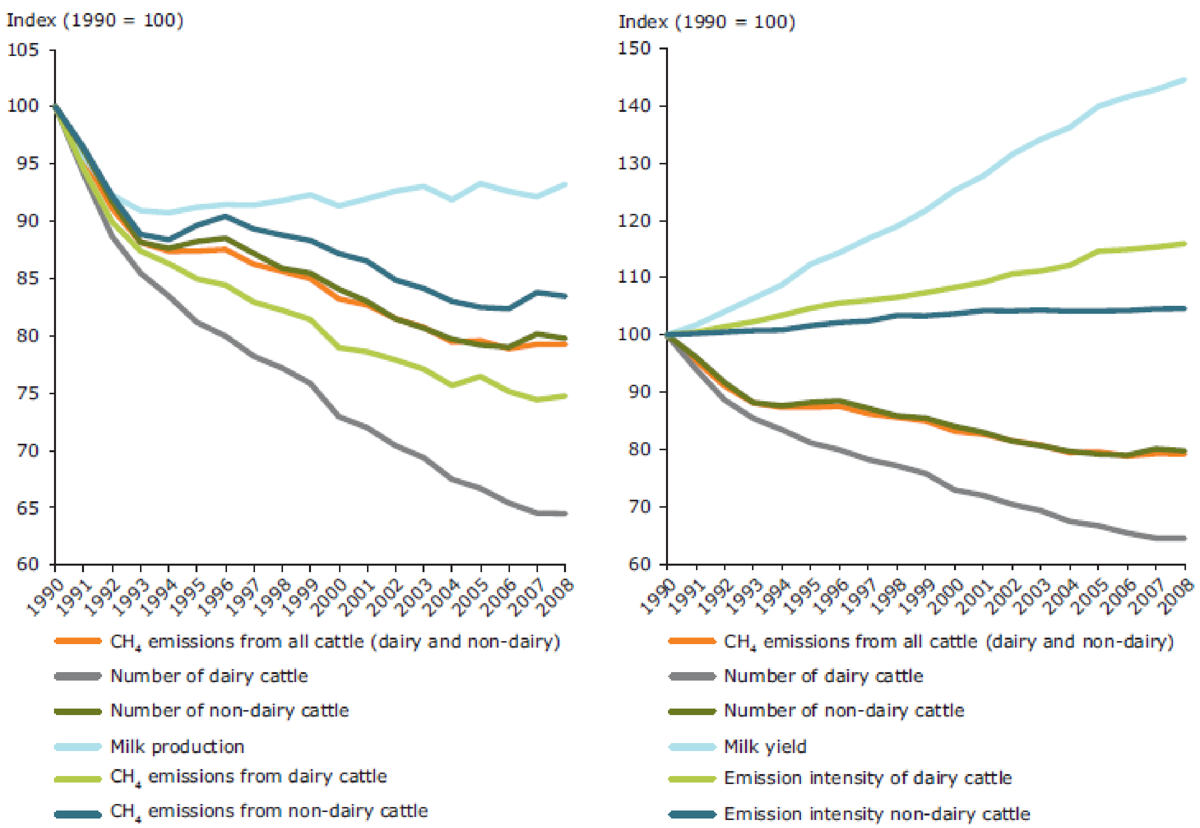


Fig. 8 - Drivers of CH_4 emissions from enteric fermentation of cattle in the EU, 1990–2008 (EEA, 2011b).

An important driver of GHG emissions from agriculture were the milk quota. For example in the Netherlands, total milk production is determined mainly by EU policy on milk quota, which remained unchanged. Therefore, the effect of increased milk production per cow needed to be counteracted by decreasing the animal number of adult dairy cattle (EEA, 2011c).
Manure management is another source of methane emission but its contribution to total EU-15 GHG emissions in 2009 is less important than the enteric methane and it accounts for 0.54%, anyway between 1990 and 2009, CH_4 emissions from this source decreased by 10% (EEA, 2011c).

The Annual European Union greenhouse gas inventory 1990–2009 and inventory report (EEA, 2011c), shows an overview of the CH_4 emissions, animal population and the corresponding implied emission factors regarding dairy cattle, the data are reported in Table 2:

Tab. 2 - Variation animal number, feed intake and milk productivity, total CH_4 emissions, implied emission factor and animal waste management system at EU-15 level for the years 1990 and 2009 for dairy cattle (EEA, 2011c).

	1990	2009	2009 value % of 1990
Animal population [1000 heads]	26210	17810	68%
Feed intake [MJ/head/yr]	248	301	121%
Milk productivity [kg/day/head]	13	18	138%
Enteric emission (1):			
Implied emission factor [kg CH4/head/yr]	100	119	119%
CH_4 enteric emission [Gg CH_4]	2627	2124	81%
Manure management emission (2):			
Implied emission factor [kg CH_4/head/yr]	18	23	128%
CH_4 manure management [Gg CH_4]	473	410	87%
Animal waste managemrnt system(3):			
Liquid system	32%	39%	
Daily spread	3%	3%	
Solid storage and dry lot	35%	29%	
Pature range and paddock	29%	28%	

(1) Dairy cattle contribute for the 36% of CH_4 enteric emission of livestock sector (includes non-dairy cattle 48% and sheep 11%).
(2) Dairy cattle contribute for the 21% of CH_4 manure management emission of livestock sector (includes non-dairy cattle for 27% and swine for 46%).
(3) Allocation of Animal Waste Management Systems (%).

Direct N_2O emissions from agricultural soils is the largest source category of N_2O emissions and accounts for 2.6% of total EU-15 GHG emissions in 2009 (EEA, 2011c).

In 2008 N_2O emissions from agricultural soils, covered 28% of total GHG emissions from the EU-27 agriculture sector. Between 1990 and 2008, N_2O emissions from agricultural soils were significantly reduced due to the lesser use of fertilizer per cropland, combined with a decreasing cropland area. Various national and EU policies aimed at reducing the amount of synthetic fertilizers applied to agricultural soils contributed to this decrease, in particular the Nitrates Directive. Its impact was the largest in the reduction of synthetic fertilizer application (primarily in the early 1990s and to a lesser extent between 2000 and 2008), but it also contributed to reducing input of organic fertilizers (EEA, 2011b) (Figure 9).

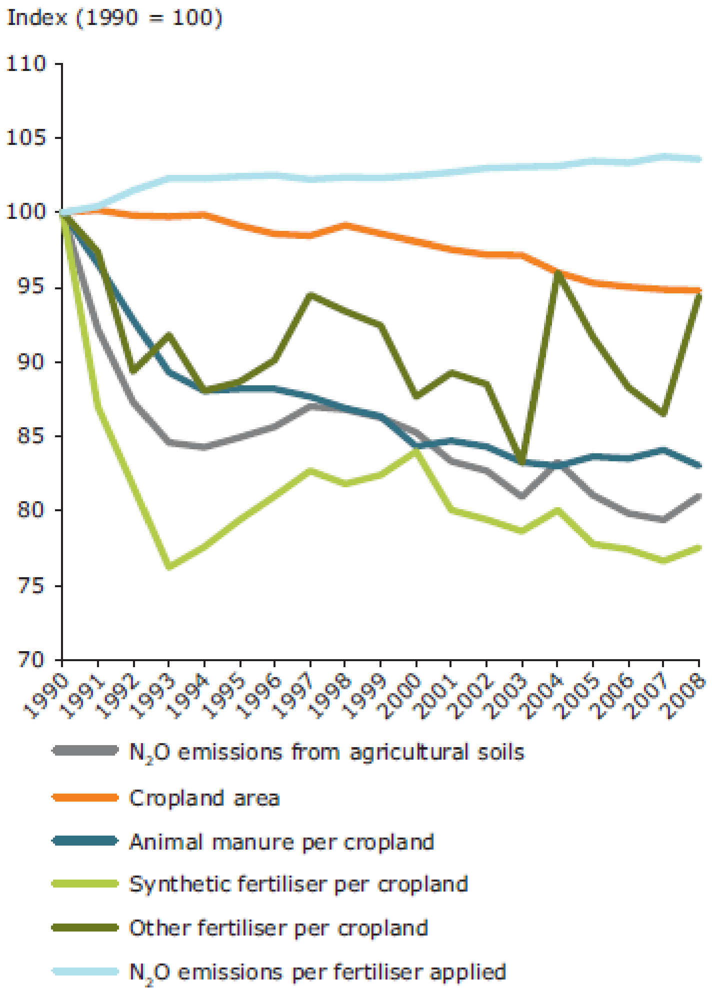


Fig. 9 Drivers of N_2O emissions from EU agricultural soils, 1990–2008 (EEA, 2011b).

As it is reported in Table 3, N_2O emissions from solid storage and dry lot account for 0.43% of total EU-15 GHG emissions in 2009. Between 1990 and 2009, N_2O emissions from this source decreased by 18%.

N_2O emissions from pasture, range and paddock manure account for 0.7% of total EU-15 GHG emissions in 2009. Between 1990 and 2009, N_2O emissions from this source decreased by 16%.

N_2O emissions from indirect emissions account for 1.75% of total EU-15 GHG emissions in 2009. Between 1990 and 2009, N_2O emissions from this source decreased by 21% (EEA, 2011c).

Tab. 3 -Total N_2O emissions, total nitrogen input into agricultural soils and implied emission factor at EU-15 level in 2009 and 1990 and relative changes (EEA, 2011c).

1990	Synthetic Fertilizer	Animal Wastes appl.	Cultiv. of Histosols[1)]	Animal Production	Atmospheric Deposition	Nitrogen Leaching and run-off
	Direct				Indirect	
Total Emissions of N_2O [Gg N_2O]	199	79	25	98	48	216
Total Nitrogen input [Gg N]	10244	4287	21011	3006	3082	5561
Implied Emission Factor [kg N_2O-N / kg N]	1.23%	1.17%	7.6	2.07%	1.00%	2.47%

2009	Synthetic Fertilizer	Animal Wastes appl.	Cultiv. of Histosols[1)]	Animal Production	Atmospheric Deposition	Nitrogen Leaching and run-off
	Direct				Indirect	
Total Emissions of N_2O [Gg N_2O]	137	74	25	82	38	172
Total Nitrogen input [Gg N]	7078	3857	20960	2619	2407	4425
Implied Emission Factor [kg N_2O-N / kg N]	1.23%	1.22%	7.7	2.00%	1.00%	2.48%

2009 value in percent of 1990	Synthetic Fertilizer	Animal Wastes appl.	Cultiv. of Histosols	Animal Production	Atmospheric Deposition	Nitrogen Leaching and run-off
	Direct				Indirect	
Total Emissions of N_2O	69%	94%	101%	84%	78%	80%
Total Nitrogen input	69%	90%	100%	87%	78%	80%
Implied Emission Factor	100%	104%	102%	97%	100%	100%

Source of information: Tables 4.D for 1990 and 2009, submitted in 2011

1) Histosols unit AD: km^2; Unit for IEF: kg N_2O-N/ha

Only few studies report a detailed assessment of the global greenhouse gas emissions from livestock sector in Europe, the most recent are Lesschen et al. (2011) and Weiss and Leip (2012) because many study focused only on environmental impact of single animal product (i.e. kg of milk, kg of beef, etc.). Both of them considered several animal products at the farm gate (i.e: milk, beef, pork, eggs plus sheep and goat milk and meat in Weiss and Leip (2012)) but a comparison between the results of the two works should be done carefully because the models used in the estimation and some basic assumptions (i.e. accounting of emissions related to LUC) can differ significantly. Anyway both studies provide an overview of how and how much the livestock sector of the European countries can affect the environment (in a global warming perspective).

Lesschen et al. (2011) found that on average, 72% of the total land area utilized for agriculture (i.e. 188 million ha) was used for animal feed and forage production (Figure 10).

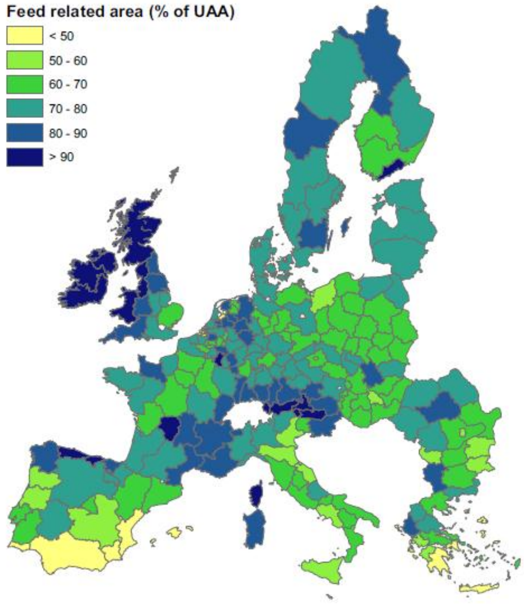


Fig. 10 - Greenhouse gas emissions for each animal product, European average and national assessment (Lesschen et al., 2011).

Total GHG emissions from livestock farming in the EU-27 were 493 Tg CO_2-eq yr^{-1}, which corresponds to about 10% of total EU-27 GHG emissions. Beef had the highest emission with 22.6 kg CO_2-eq kg^{-1}, followed by pork at 3.5 kg CO_2-eq kg^{-1}, eggs at 1.7 kg CO_2-eq kg^{-1}, poultry at 1.6 kg CO_2-eq kg^{-1} and milk at 1.3 kg CO_2-eq kg^{-1} for the EU-27. Summarized by sector, the largest livestock related GHG emissions in the EU were from dairy followed by beef (Figure 11). Together, these sectors account for more than 70% of GHG emissions from livestock production. The GHG emission from the pig sector was about 16%, whereas the poultry sectors was about 6%.

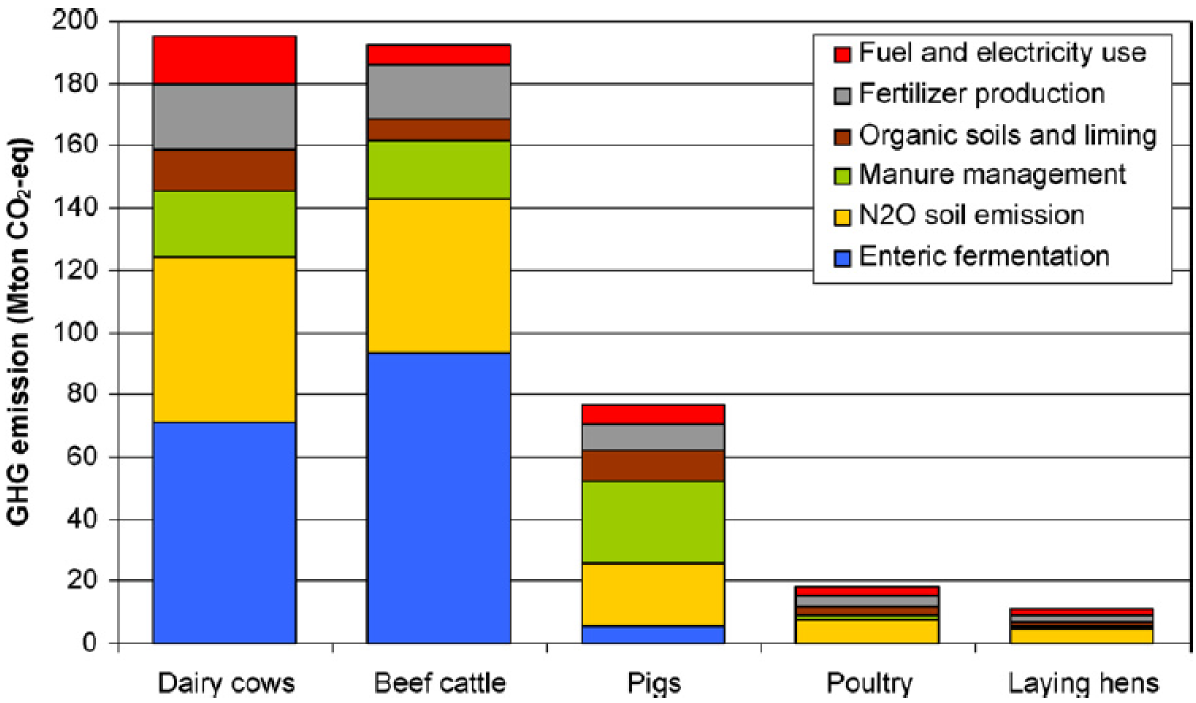


Fig. 11 - Total greenhouse gas emissions from the various emission sources associated with livestock production in the EU-27 (Lesschen et al., 2011).

The two GHG emission sources which have relatively large contributions are CH_4 emissions from enteric fermentation at 36% and N_2O emission from soils at 28%. GHG emissions from manure storage accounted for 13%, fertilizer production 11%, cultivation of organic soils and liming 7%, fossil fuel use 3.2% and electricity 3.2% of total GHG emissions from livestock production.
The contribution of different sectors of livestock production to total agricultural greenhouse gas emissions differs substantially among countries (Figure 12).

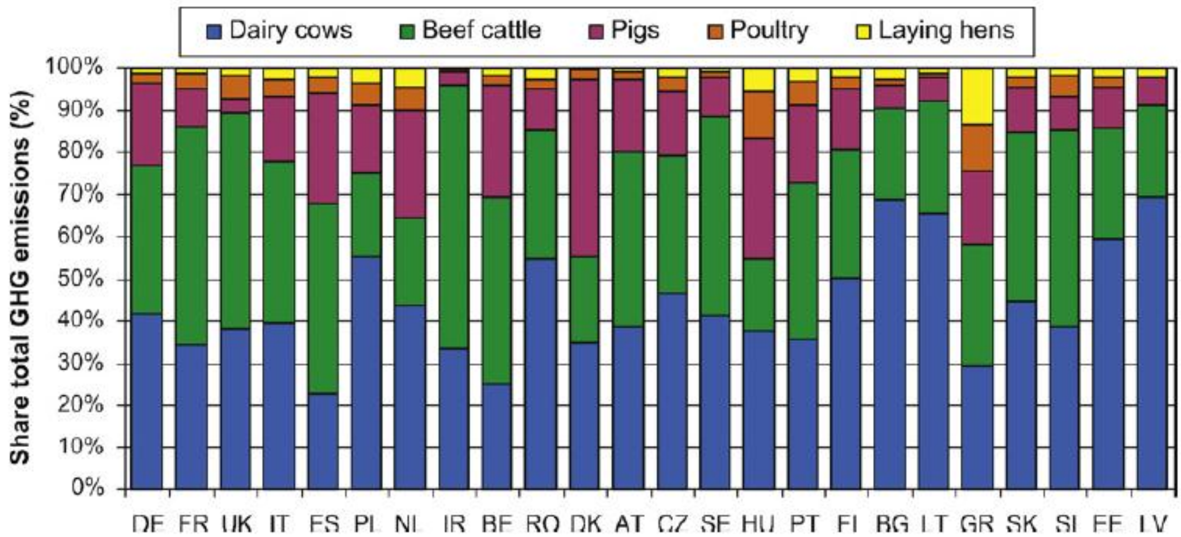


Fig. 12 - Share of the different sectors of livestock production in total agricultural greenhouse gas emissions from each country. The countries on the *x-axis* are ordered according to the magnitude of greenhouse gas emissions (Lesschen et al., 2011).

Moreover, considering only a single sector (i.e. milk production), the variation in terms of total emission per kg of product and contribution of each compartment to this emission is wide among the countries (Figure 13).

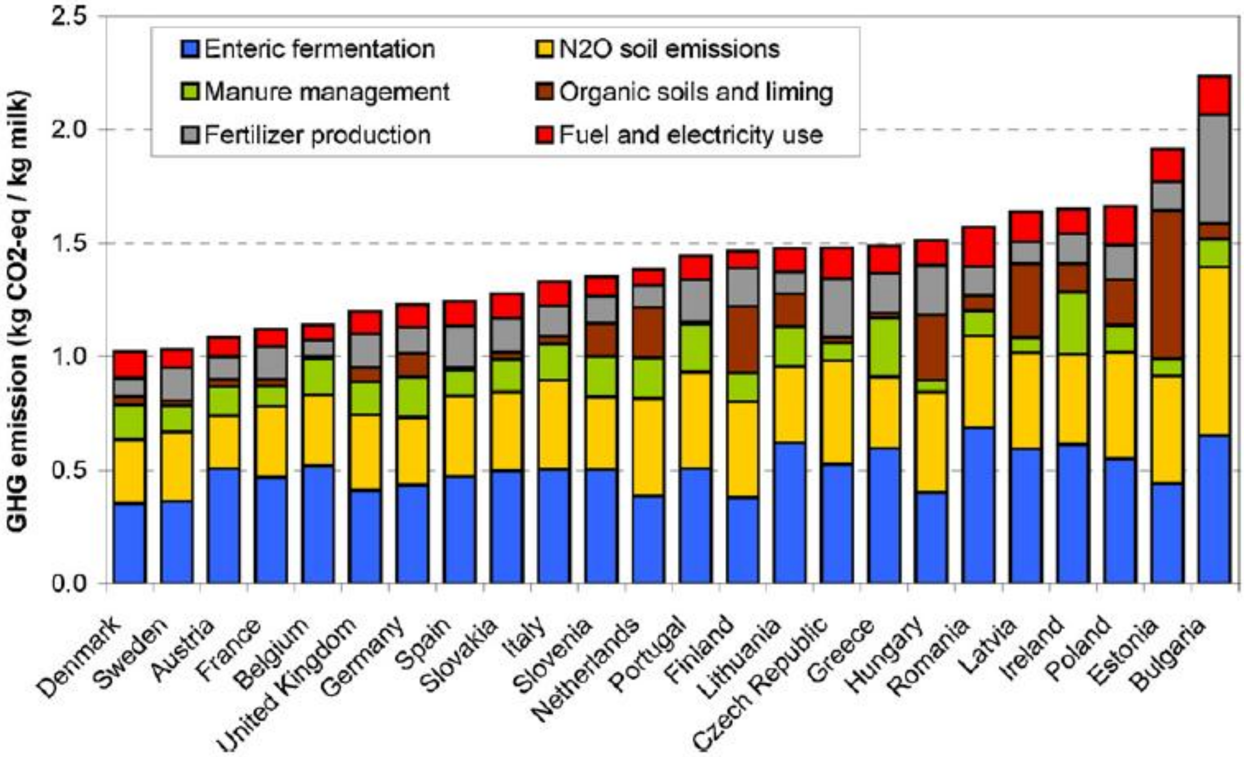


Fig. 13 - GHG emission per kg milk within EU countries as it relates to emission sources (Lesschen et al., 2011).

Weiss and Leip (2012) found that, on product level, the total GHG intensities of ruminants amount to 19-28 kg CO_2 eq. per kg of meat (21-28 kg for beef and 19-28 kg per kg of sheep and goat meat) on EU average, while the production of pork (7-10 kg) and poultry meat (5-7 kg) creates significantly less net emissions. GHG fluxes per kg of cow milk are estimated at 1.3-1.7 kg CO_2 eq. on EU-27 average, those from sheep and goat milk at 2.6-4.1 kg CO_2-eq. kg^{-1} product. The production of eggs leads to the net emission of 2.8–3.2 kg of CO_2 eq. per kg of eggs on EU average. The variation in greenhouse gas emission in each animal product is depending on which scenario is used for the calculation of LUC emissions. In the following pictures (Figure 14) are represented the impacts estimated for the different animal products for every country considered in the study. As seen before in the results of Lesschen et al. (2011) the estimated impact (per 1 kg of product) and the contributions to this impact can differ substantially among the countries.

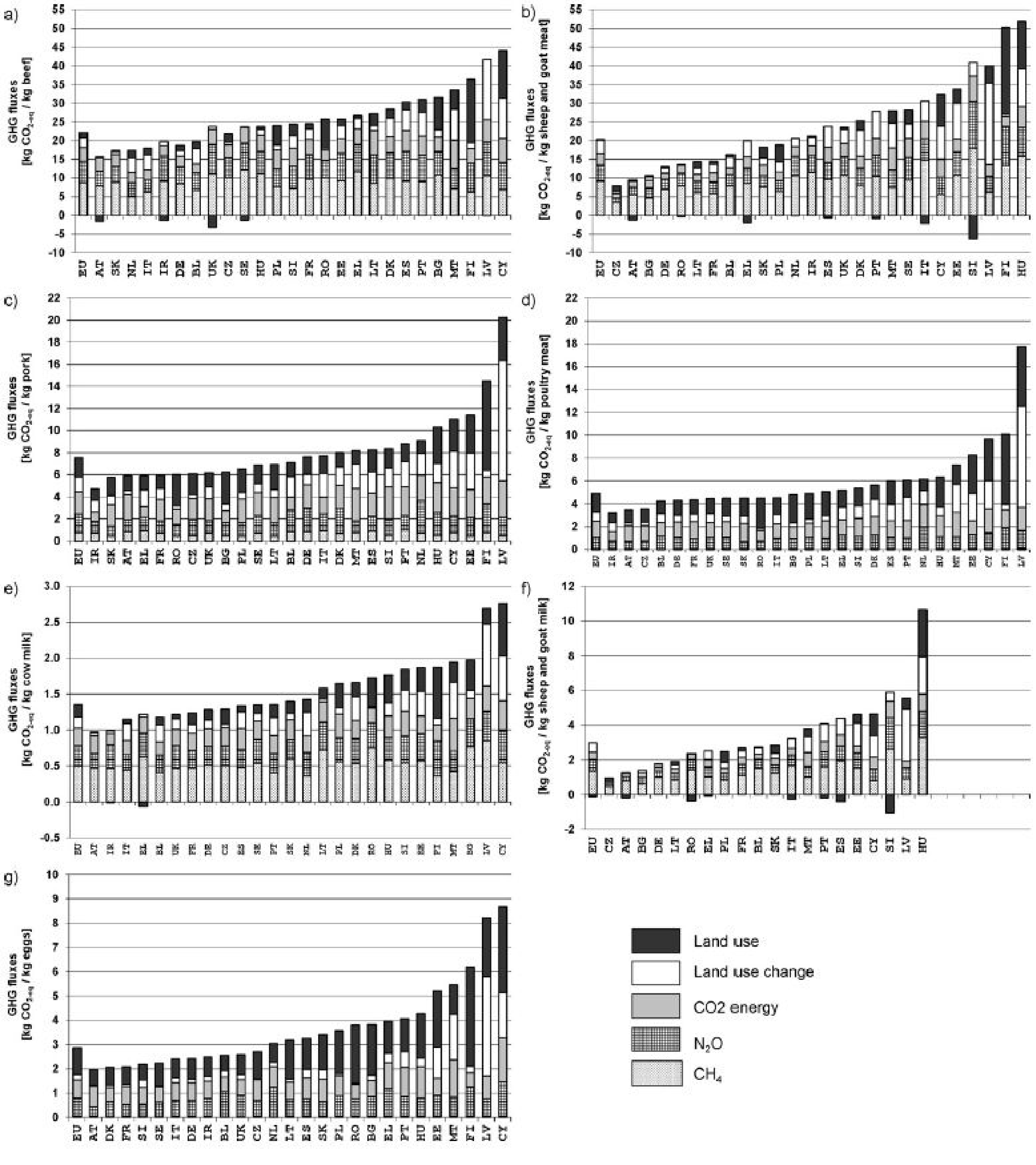
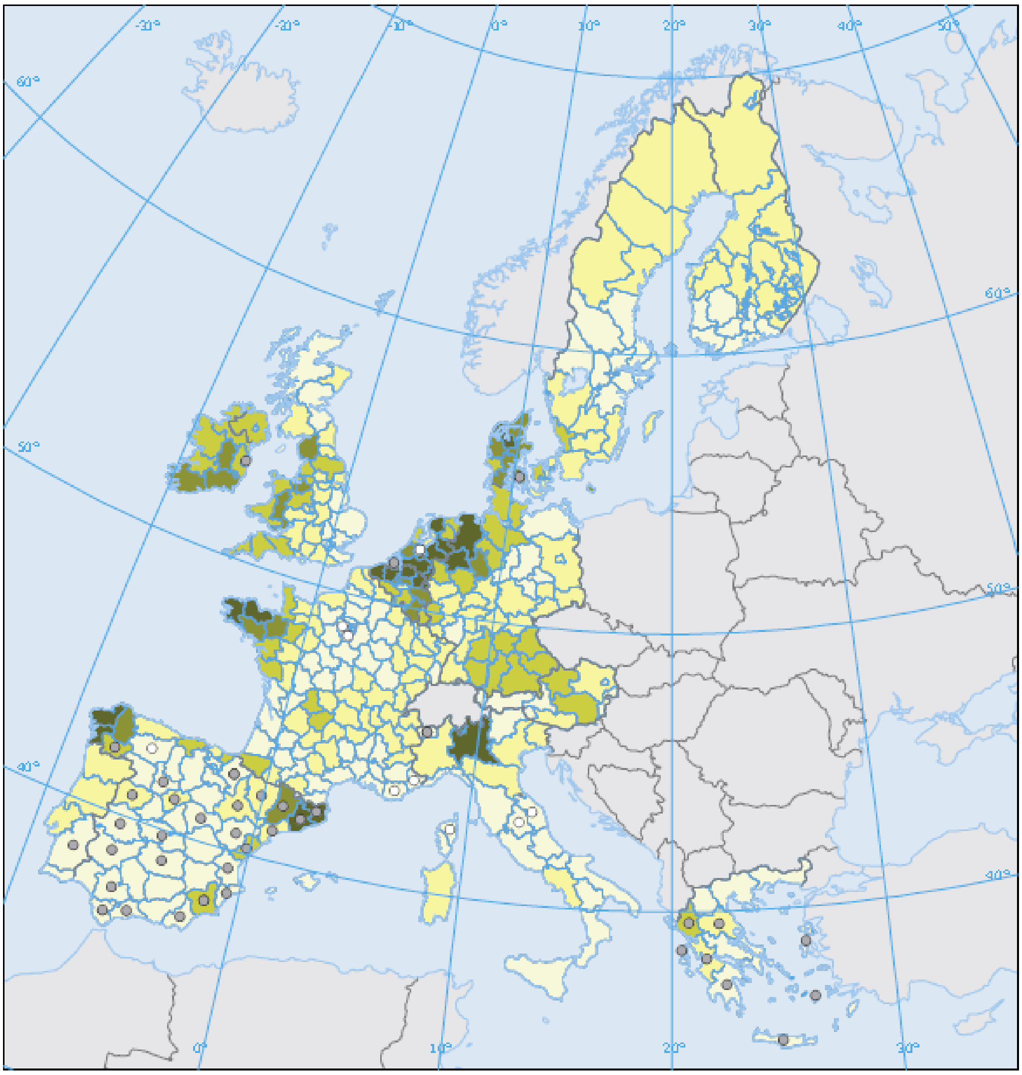


Fig. 14 - Greenhouse gas emissions for each animal product, European average and national assessment (Weiss and Leip, 2012).

1.3.2. Nitrates losses and water quality

Environmental legislation is an important policy tool for protecting water quality (EAA, 2005). The Nitrates Directive (Council Directive 91/676/EEC – E.C., 1991) aims at reducing and preventing water pollution caused by nitrates from agricultural sources with the goal that nitrate concentrations in groundwater will not exceed 50 mg NO_3 l^{-1} and listing codes of good practice to be implemented by the farmers on a voluntary basis. Nitrate vulnerable zones must be designated on the basis of monitoring results which indicate that the groundwater and surface waters in these zones are or could be affected by nitrate pollution from agriculture. The action program must contain mandatory measures relating to: (i) periods when application of animal manure and

fertilizers is prohibited; (ii) capacity of and facilities for storage of animal manure; and (iii) limits to the amounts of animal manure and fertilizers applied to land (EEA, 2011c).
The Nitrates Directive limits the land application from livestock manure to 170 kg N ha^{-1} per year in designated zones to which action programs apply. This application standard is established in almost all action programs. A further increase in the area of vulnerable zones as compared to the former reporting period is observed in the EU 15. Designated zones increased from 43.7 to 44.6 % of the EU 15 territory, while 40.9 % of the EU 27 territory is designated, including the territory of Member States that apply an action program on the whole territory. The Nitrates Directive allows for the possibility for a derogation in respect to the maximum amount of 170 kg nitrogen per hectare per year from livestock manure, provided that it is demonstrated that the directive's objectives are still achieved and that the derogation is based on objective criteria such as long growing seasons, crops with high nitrogen uptake, high net precipitation or soils with a high denitrification capacity (E.C., 1991).
A report from the European Commission (E.C., 2011a) concerning the protection of waters against pollution caused by nitrates from agricultural sources based on Member State reports for the period 2004-2007, shows that the progressive reduction in mineral nitrogen fertilizer consumption, which started in the early 1990s, stabilized during the period 2004-2007 for the EU 15. At EU 27 level the nitrogen consumption shows a slightly increasing trend. As compared to the last reporting period, the yearly total amount of mineral nitrogen fertilizer consumption remained stable around 9 million tons in the EU-15 whereas it has increased by 6%, from 11.4 to 12.1 million tons, in the EU 27.
The amount of nitrogen from animal husbandry spread annually on agricultural soils in the EU 27 has decreased from 9.4 to 9.1 million tons between 2003 and 2007 and from 7.9 to 7.6 for the EU15. There are large differences in pressure from agriculture between Member States. Areas with a high nutrient pressure include among others the Netherlands, Belgium-Flanders and France-Brittany. Member States in Eastern Europe generally have lower pressures due to lower input of fertilizers and livestock density (E.C., 2011a).



(*) NUTS 2: Austria, Belgium, Germany, Greece, Italy, Luxembourg, Netherlands, Portugal and the United Kingdom.
NUTS 3: Denmark, Finland, France, Ireland, Spain and Sweden.

NUTS = Nomenclature of territorial units for statistics.
© EuroGeographics Association for the administrative boundaries.

Note: Poultry figures are part of the calculation of national gross nitrogen balances but not included in this graph. Adding poultry production would emphasize some regional livestock hot spots, for example in the Benelux region.

Source: Community survey on the structure of agricultural holdings (FSS), Eurostat.

Fig. 15 - Regional distribution of cattle, sheep and pig livestock units (LU) per ha of UAA in 2000 and change from 1990–2000 (EAA, 2005).

The contribution of nitrogen loads from agriculture to surface waters is decreasing in many Member States. Nevertheless, the relative contribution from agriculture remains high. In most Member States agriculture is responsible for over 50% of the total nitrogen discharge to surface waters.

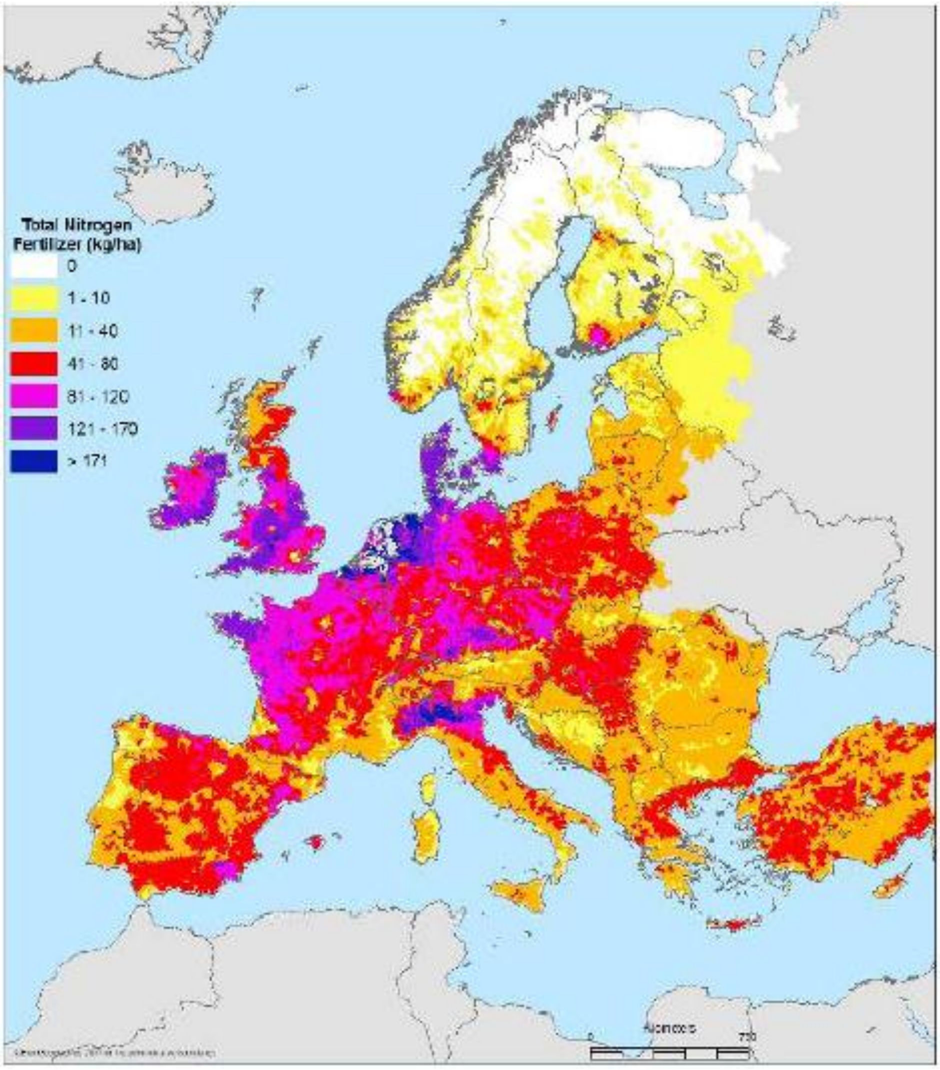


Fig. 16 - Total nitrogen application, manure and chemical fertilizer for year 2005 (E.C., 2011a).

Regarding water quality, for groundwater, 66% of the monitoring stations show stable or decreasing nitrate concentrations. However, in 34% of the stations an increase in nitrate pollution is still observed and 15% of stations show nitrate concentrations above the quality threshold of 50 mg l^{-1}. Within groundwater bodies, shallow levels show higher nitrate concentrations than deeper levels. The highest proportion of contaminated water lies between 5 and 15 meters below the surface.

For fresh surface water, 70% of the monitoring stations show stable or decreasing nitrate concentrations. In 3% the concentration is exceeding 50 mg l^{-1} while in 21% the concentration is below 2 mg l^{-1}. In 33% of the stations monitoring trophic status, the water is defined eutrophic or hypertrophic

Phosphorus concentrations in some EU rivers have fallen since the mid-1980s, particularly in the largest and most polluted rivers. Although phosphorus pollution from point sources has fallen, it may be necessary to take measures to reduce diffuse loads from agricultural areas — particularly in areas where the soil's absorption capacity for phosphorus is exceeded (EAA, 2005).

Consumption of mineral phosphorus fertilizers reduced with 9% for EU the 15, while it decreased by only 1% for the EU 27 as compared with the last reporting period (E.C., 1991).

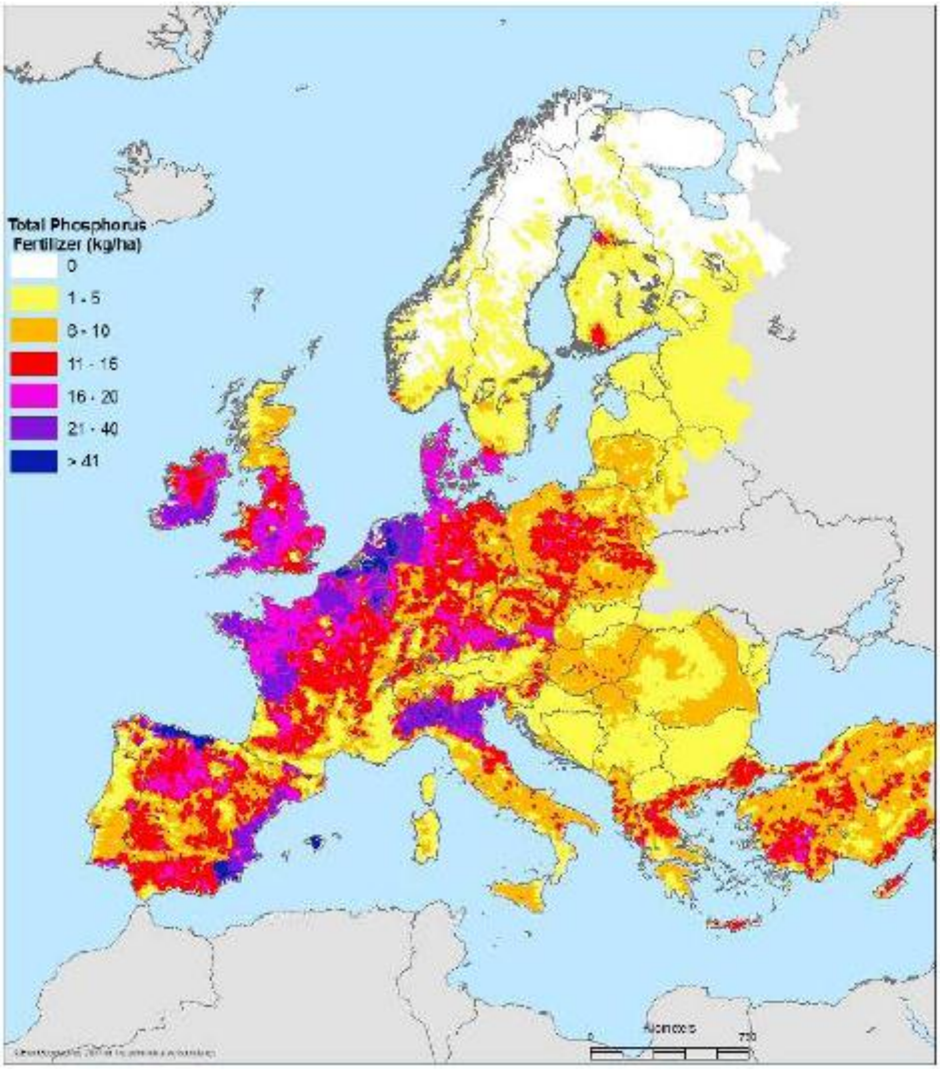


Fig. 17 - Total phosphorus application, manure and chemical fertilizer for year 2005 (E.C., 2011a).

1.3.3. Ammonia emissions from agricultural activities and acidification effects

In general, the EU Member States have made excellent progress in reducing emissions that decreased by 26% between 1990 and 2009. The agricultural sector remains the major source of NH_3 emissions (94% of total 2009 emissions) (EAA, 2011a).

These emissions derive mainly from the decomposition of urea in animal wastes and uric acid in poultry wastes (EAA, 2011a).

Emissions depend on the animal species, age, weight, diet, housing systems, waste management and storage techniques. The majority of the reduction in emissions is due to the combination of reduced livestock numbers across Europe (especially cattle), and the lower use of nitrogenous fertilizers (EAA, 2011a). The Nitrates Directive affected emissions in most countries, for example in Belgium, manure Action Plans (based on the Nitrate directive) in Flanders affected NH_3 volatilization from manure application. The first action plan in 1991 regulated the reduced in which manure can be spread and foresees low emission techniques for the application of manure on land (EEA, 2011c).

1.4. THE LOMBARDY CONTEXT

The Lombardy Region has a territory with a surface about 2,380,000 ha of which approximately 1,000,000 is dedicated to an agricultural use and the 80% of that is flat land. Crops production for livestock feed occupies almost the 80% of the agricultural land (AL). The 36% of the AL is dedicated to grow maize (mainly for grains and less for silage), this crop is more located in flat land especially in the zones with high livestock rate. Permanent meadows and pastures occupy almost the 23% of the AL and they are mainly located in mountain and hilly area. The 7% of the AL is used to produce other forages (generally crops in rotation) and they are located primarily in flat land. Almost the 14% of the AL is dedicated to winter cereals production, mainly wheat and then barley, rye, oats, etc.

In 2008 the total number of cattle reared in Lombardy was 1,535,840 spread on 15,249 farms (this data is referred to September 2009). The regional pig sector counts 4,820,489 heads in 2008 and it represents more than 52% of the national production. The 72% of regional production is located in the flat land (ERSAF, 2009).

About the dairy production in Lombardy, in 2011 the total number of dairy cows reared was 543,179 (the 31% of all national consistency) with an average production of 9,242 kg of milk cow^{-1} lactation^{-1} (CLAL, 2012). In 2011 the amount of milk delivered to the dairy industry was 4,452,814 tons about the 41% of national milk production (CLAL, 2012).

The data presented before show that in this region livestock production makes an high pressure on the environment, generally the high number of animals bred doesn't match an availability of sufficient land to cover all the requirements in term of feed, for that reason farms with high livestock rate show a productive system with high external inputs in terms of commercial feed and roughages and low output (milk and meat). If an animal production system import most animal feed from elsewhere, while manure is often not transported back, this prevents nutrient recycling (Naylor et al., 2005). This situation is exacerbated if it is considered that in this areas is also used a great amount of fertilizer in crops production (especially nitrogen) despite the high availability of manure from livestock farms. This unbalanced situation between input and output creates some environmental problems like nitrate leaching and ammonia emissions. However, farmers, animal feed companies and meat processing industries invest in a large specialized animal production systems because of their high productivity/unit labor, capital and land. These systems benefit from economies of scale, specialization and intensification (Roberts, 2008).

In the 1991 the European Commission issued the Nitrates Directive (Council Directive 91/676/EEC – E.C., 1991) which limits application of animal manure N in 'nitrate vulnerable zones' to a maximum of 170 kg ha^{-1} yr^{-1}, corresponding to 1.7 livestock units per ha (LU ha^{-1}) (1 LU is the relative weight of a mature dairy cow). Large pig and poultry farms require a permit (i.e. 'license to produce') and must adopt best available techniques (BAT) prescribed by IPPC Directive 2008/1/EC (E.C., 2008). In 2006 the Regional Commission (D.g.r., 2006) defined the vulnerable zones (with a maximum amount of 170 kg ha^{-1}) and the non-vulnerable zones (with a maximum amount of 170 kg ha^{-1}). In Lombardy the 56% of flat lands and the 62% of agricultural surface located in flat land are respectively in vulnerable zones (Figure 18). In September 2009 the total number of livestock farms (cattle, swine and poultry) in Lombardy was counted about 20,106 units and of these 9,674 (48%) are in nitrate vulnerable zones and 10,432 (52%) are in non-vulnerable zones (ERSAF, 2009).

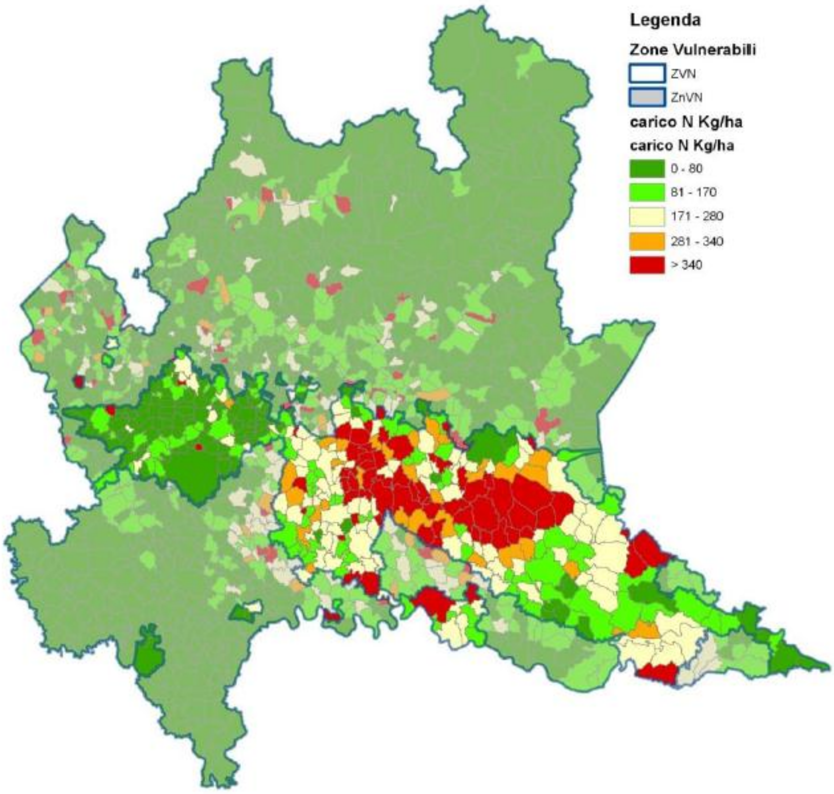


Fig. 18 - Nitrogen load on field (kg ha^{-1}) (SIARL, 2009).

The cattle herds mainly contribute to regional nitrogen production with a share of 60% of annual amount, followed by swine sector with the 28% of the total and the poultry sector with 11%, the other 1% is related to the other animal categories.
If is considered the vulnerable area in the 5 most important province for the livestock production in Lombardy the situation is very critic. The limit of 170 kg N ha^{-1} in the vulnerable zones in all cases is exceeded (Table 4).

Tab. 4 - (ERSAF, 2009)

N in NON VULNERABLE ZONES				
Province	Amount of N in field (kg/NVZ)	Agricultural land (ha NVZ)	Amount of N in field (kg/ha NVZ)	Missing land (ha)[a]
BERGAMO	9,697,764	30,119	322	-26927
BRESCIA	34,634,250	108,351	320	-95380
CREMONA	19,027,567	75,512	252	-36415
LODI	3,360,369	15,656	215	-4111
MANTOVA	24,859,271	126,716	196	-19515

[a] land required to cover the nitrogen surplus when 170 kg N ha^{-1} is the maximum amount permitted.

The 3rd November 2011 the European Commission approved the derogation (E.C., 2011b) to the Nitrate Directive 91/676 for the some Italian Region in which Lombardy. This derogation makes possible to rise the amount of nitrogen per hectare in vulnerable zones from 170 kg to 250 kg. But the farms want to join this derogation need to improve the agronomic manure management, to spread manure mainly in spring season with low emission techniques on some crops with long growing season (they should be at least the 70% of farm land) like permanent grass (with less than 50% of legumes), maize FAO class 600-700 (sown at the end of March or at the beginning of April)

with at least 145-150 days of growing season and all the plants should be harvested or removed from the field, maize or sorghum followed by a winter crop, winter cereal (like wheat or barley) followed by a summer crop. Moreover is allowed to use cattle slurry and manure and only the clarified fraction of swine slurry and at least two third of the total amount of nitrogen should be used before the 30th of June of every year (Bonazzi and Mantovi, 2011). Remain out of the derogations poultry farms.
In this context is clear that the general attention is mainly focused to limit the environmental pressure on a local scale (especially on nitrate leaching and ammonia emissions) but a global evaluation should be necessary to better identify which are the hot spots of the environmental impact of a product chain.

1.5. LCA METHODOLOGY APPLIED TO MILK PRODUCTION AT THE FARM LEVEL

1.5.1. General overview

There are increasing concerns about the ecological footprint of animal production. As described before livestock production systems have been linked to expansion of agricultural land and associated deforestation, emissions of the greenhouse gases (GHG), eutrophication of surface waters and nutrient imbalances (Lesschen et al., 2011). There is an increased global demand for dairy products despite bovine milk production being criticized for its environmental impacts such as greenhouse gas emissions, eutrophication and soil degradation. Increased intensification has exacerbated environmental impacts (Yan et al., 2010).
LCA is an assessment tool that "addresses the environmental aspects and potential environmental impacts throughout a product's life cycle from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal, i.e. "cradle to- grave" (ISO, 2006a).
In recent years, researchers from European countries where animal husbandry is important (e.g. France, Germany, Ireland, the Netherlands, Sweden, and UK) have applied LCA to milk production in response to environmental impact concerns (Yan et al., 2010).
Life Cycle Assessment was originally applied to analyze industrial process chains, but has been adapted over the last 15 years to assess the environmental impacts of agriculture. The LCA method involves the systemic analysis of production systems, to account for all inputs and outputs associated with a specific product within a defined system boundary (Gerber et al., 2010).
The life cycle assessment (LCA) procedure has gained prominence over the past decade as the method of choice when measuring GHG and other environmental impacts of a product, process, or service. The International Organization for Standardization (ISO) has outlined the principles and framework found in ISO 14040 (ISO, 2006a) and requirements and guidelines found in ISO 14044 (ISO, 2006b). As outlined with the ISO 14040 standard, there are 4 phases of an LCA (Milani, 2011):

- goal and scope definition
- inventory analysis
- impact assessment
- interpretation

The main strengths of LCA lie in its ability to provide a holistic assessment of production processes, in terms of resource use and environmental impacts, as well as to consider multiple parameters (Gerber et al., 2010).

1.5.2. Goal and scope definition

The goal, stated at the beginning of the project, includes objectives, intended audience and application (ISO, 2006a).

The scope includes the production system, the functional unit (FU), the system boundary, allocation procedures and any other relevant factors (ISO, 2006a).
The system boundary largely depends on the goal of the study (Gerber et al., 2010) and it determines which unit processes are included in the LCA.
The assessment can encompasses the entire production chain of cow milk, from feed production through to the final processing of milk and meat, including transport to the retail sector. The cradle to retail system boundary is split into two sub-systems:
1. Cradle to farm-gate includes all upstream processes in livestock production up to the point where the animals or products leave the farm, i.e. production of farm inputs, and dairy farming.
2. Farm-gate to retail covers transport to dairy plants, dairy processing, production of packaging, and transport to the retail distributor.
An example of system boundary, "cradle to farm gate", used in a milk LCA study is represented below (Figure 19):

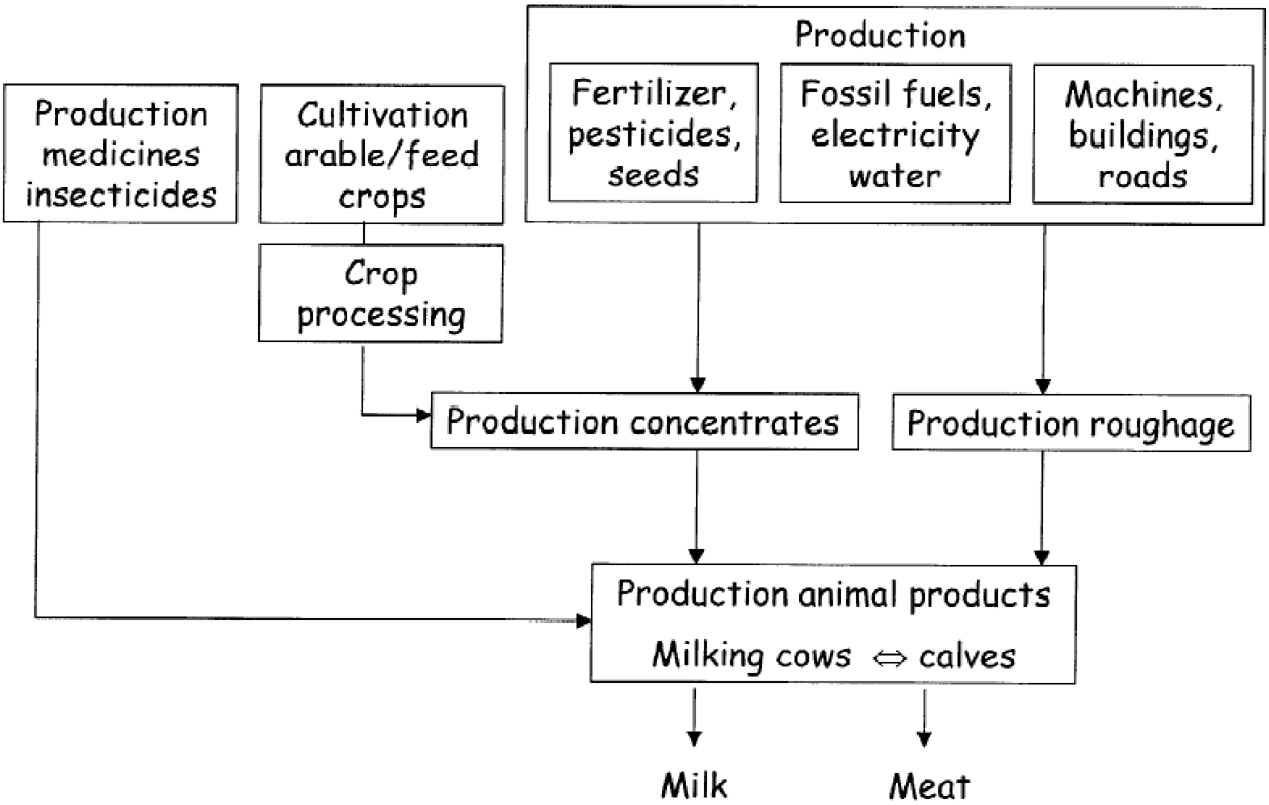


Fig. 19 – System boundaries of "cradle to farm gate" life cycle assessment (de Boer, 2003).

Most European milk LCA studies (Cederberg and Mattsson, 2000; Haas et al., 2001; de Boer, 2003; Hospido et al., 2003; Thomassen et al. 2008a; É.G. Castanheira et al., 2010; Müller-Lindenlauf et al., 2010; D. O'Brien et al., 2012a) have considered neither the entire life cycle of the product, and have focused only on the farm or dairy unit. Therefore, they are "partial LCAs" (Yan et al., 2010). This is perhaps appropriate because 80% of the GHG emissions and 40% of the energy use associated with milk are due to production (IDF, 2010). European milk LCAs have tended to use "cradle-to farm- gate" (de Boer, 2003) and include on-farm processes, off-farm production of feed, fertilizers, energy and their transportation.
In order to compare systems, a functional unit (FU) is needed. The FU describes the primary function fulfilled by a product system and enables different systems to be treated as functionally equivalent (Guinée et al., 2002).

The most common approach was modified milk mass. Swedish and Irish scientists used “1 kg energy corrected milk (ECM)” (Sjaunja et al., 1990):

kg ECM = kg milk x (0.25 x 0.122 x Fat % + 0.077 x Protein %)

while Dutch scientists used “1 kg fat and protein corrected milk (FPCM)”:

kg FPCM = kg milk x (0.337 + 0.116 x Fat % + 0.06 x Protein %)

Both Equations and can be seen as quality-corrected FU (Yan et al., 2010).

As many agricultural production systems, dairy-cattle production systems produce a mix of goods and services like edible products (milk and meat) and non-edible products and services (draught power, leather, manure, fodder, grains and capital). For that reason it is necessary to attribute environmental impacts to each product from the system using an allocation approach. Allocation describes how “inputs” and “outputs” are partitioned between the product of interest and by-products (ISO, 2006b).

In most milk LCA studies the allocation procedure considers only milk and meat as final co-products, but the method to weight the environmental impact of these two products can widely differs.

The allocations procedure described by ISO 14044 (2006b) suggests:

1- Wherever possible allocation should be avoided by
    a. dividing the unit process to be allocated into two or more sub-process and collecting the input and output data related to these sub-processes, or
    b. expanding the product system (known as system expansion) to include the additional function related to the co-products.

The basic idea of system expansion is that there is an alternative way of generating the exported functions, i.e. the co-products. Exported functions are here defined as functions that are generated in the product life cycle studied but utilized in another product life cycle. If data are available for the alternative production of the co-products, the boundaries of the systems investigated can be expanded to include the alternative production of exported functions (Cederberg and Stadig, 2003). In these work the authors reported that milk output per cow increased over the recent decade and a corresponding decrease occurred in the number of dairy cows. This would seem to be an advantageous gain relative to environmental impact for the dairy industry, but an increase in the number of beef head occurred to compensate for the beef demand (Yan et al., 2010). Apart from handling the allocation problem, system expansion makes it possible to model the indirect actions, i.e. the effects on the environmental burdens from activities outside the boundaries of the life cycle investigated (Cederberg and Stadig, 2003).

In some milk LCA studies (Haas et al., 2001; Casey and Holden, 2005a; Van der Werf et al., 2009) the environmental impact was expresses on 1 ha of land.

2- Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products of functions in a way that reflects the underlying physical relationship between them, they should reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system.

    An examples of physical allocation are:
    - Allocation according to the proportions of milk and meat protein produced, as recommended by (Gerber et al., 2010).
    - Biological allocation, based on feed energy required to produce the amount of milk and meat at the farm, was made by the empirical relation developed by IDF (2010) from data representing a large variation in type of feed rations, proportion of meat and type of animals.

3- Where physical relationship alone cannot be established or used as basis for allocation, the inputs should be allocated between the products or functions in a way that reflects other relationship between them. For example, input and output data might be allocated between co-products in proportion of the economic value of the products.

Economic allocation was used by several authors in milk LCA studies (Cederberg and Stadig, 2003; Hospido et al., 2003; Cederberg and Flysjö, 2004; Casey and Holden, 2005a; Casey and Holden, 2005b; Van der Werf et al., 2009; Thomassen et al., 2008a; Thomassen et al., 2008b; Thomassen et al., 2009).

Looking at the whole life cycle of milk and dairy products from farm to manufacturing gate out, there are several process that involve multiple co-products. If we consider a partial life cycle assessment ("from the cradle to farm gate") the production of feed is one of the process that generate more than one product, and therefore the environmental burden should be distributed between the co-products.

Some of the more commonly used feed ingredients for dairy cows where allocation situation occurs are:

- Soy meal (co-product to soy oil and soy hull, produced from soy beans)
- rape seed meal (co-product to rape seed oil produced from rape seed)
- palm kernel expels (co-product to palm kernel oil, produced from palm kernels, which is a co-product to palm oil, produced from oil palm)
- maize gluten meal (co-product to maize gluten feed, maize germ meal and maize starch, produced from maize)
- wheat bran (co-product from wheat flour, produced from grain)
- dry distillers grains with solubles (DDGS), co-product to corn ethanol, produced from corn grains, and
- other co-product like: beet pulp from sugar beet, molasses from sugar cane, DDGS (barley and wheat) from distillery, cotton seed delinted, etc…

The IDF guide (2010) suggests to use economic allocation for co-product in feed production. These is identified as the most feasible allocation method to use at these stage because:

- subdivision of the system is not typically possible for feed products
- it can be difficult to identify the product/s that has/have been substituted by the by-products to apply the system expansion method, and it can be time consuming
- it is difficult to find a physical relationship that reflects the relation between inputs and outputs, for example soy meal is typically used for its protein content, while soy oil is used for its energy content, hence applying allocation based on protein content or energy bases is not an allocation factor that is relevant for both products.

Consequently, economic allocation is the recommended method in this situation. As many feed ingredients are produced regionally or locally, five year average on prices are advised to minimize fluctuations between years.

1.5.3. Inventory analysis

Life cycle inventory analysis involves compilation of inputs, outputs and emissions for a product system throughout its life cycle (ISO, 2006b). The aim of this stage is to develop a model which quantifies the resources used and the amount of waste and emissions generated/functional unit (Crosson et al., 2011).

In LCI (Life Cycle Inventory), data relating to the input and output of each process are collected. For milk production, depending on the goals and scope, this can require data from farms, feed processors, dairy industries, retailers, and waste treatment. Data collecting is particularly time consuming.

1.5.4. Impact assessment

The LCIA phase consists of mandatory and optional elements (ISO, 2006b). For the mandatory elements LCI data are processed by “environmental mechanisms” into specific environmental impact categories (classification), and then characterization factors (CFs) for each category are applied to calculate a category indicator value (characterization).

The baseline impact categories are abiotic resource depletion, land use, climate change, stratospheric ozone depletion, human/eco-toxicity, photo-oxidant formation, acidification and eutrophication (Guinée, 2002).

- Depletion of abiotic resources: The issue related to the depletion of abiotic resources, such as fossil fuels or minerals is their decreasing availability for future generations (Brentrup et al., 2003). This impact category is concerned with protection of human welfare, human health and ecosystem health (Garret and Collins, 2009);
- Land use: the ‘land use’ impact category describes the environmental impacts of utilizing and reshaping land for human purposes. The environmental consequences of land use such as arable farming or urban settlement are the decreasing availability of habitats and the decreasing diversity of wildlife species (Brentrup et al., 2003);
- Climate change: Emissions of gases like carbon dioxide (CO_2) and nitrous oxide (N_2O) lead to an unnatural warming of the Earth’s surface, which in turn will cause global and regional climatic changes. This environmental impact is commonly described as ‘global warming’. The term ‘climate change’ indicates that the possible consequences of global warming concern more elements of the global climate than only the temperature (e.g. precipitation, wind) (Brentrup et al., 2003). The characterization model as developed by the Intergovernmental Panel on Climate Change (IPCC) is selected for development of characterization factors. Factors are expressed as Global Warming Potential for time horizon 100 years (GWP_{100}), in kg carbon dioxide equivalent (kg CO_2 eq.);
- Stratospheric ozone depletion: because of stratospheric ozone depletion, a larger fraction of UV-B radiation reaches the earth surface. This can have harmful effects upon human health, animal health, terrestrial and aquatic ecosystems, biochemical cycles and on materials (Garret and Collins, 2009);
- Human toxicity: This impact category includes all direct toxic effects of emissions on humans (Brentrup et al., 2003).
- Photo-oxidant formation: Photo-oxidant formation is the formation of reactive substances (mainly ozone) which are injurious to human health and ecosystems and which also may damage crops. This problem is also indicated with “summer smog” (Garret and Collins, 2009);
- Acidification: Acidifying substances cause a wide range of impacts on soil, groundwater, surface water, organisms, ecosystems and materials (Garret and Collins, 2009);
- Eutrophication: Eutrophication can be defined as an undesired increase in biomass production in aquatic and terrestrial ecosystems caused by high nutrient inputs, which result in a shift in species composition (Brentrup et al., 2003).
- Energy use: the total amount of primary energy consists of the cumulative sum of i) the direct energy due to the use of fuels and electricity, ii) the indirect energy associated with the production of materials, equipment, etc., and iii) the energy contained in any feedstocks, such as chemicals and materials derived from fossil fuels (Malça and Freire, 2004).

Generally the most common impact studied is climate change, followed by abiotic resource depletion, acidification and eutrophication. These reflect regional interest in global warming and nutrient loss (as regulated by the Nitrates Directive and other legislations) (Yan et al., 2010).
Agricultural land use affects several environmental impacts such as biodiversity, landscapes and soil quality, which are all impact categories not usually covered by LCA and with no widely accepted assessment method (Milà i Canals et al., 2007). Several attempts have been made to include land use in LCA (see Milà i Canals et al. 2007, Koellner and Scholz, 2008, Penman et al., 2010, de Baan et al., 2012 for references), but proposed indicators are in most cases not checked with a consistent framework (Milà i Canals et al. 2007).
Biodiversity is a complex and multifaceted concept, involving several hierarchical levels (i.e., genes, species, ecosystems), biological attributes (i.e., composition, structure, function) and a multitude of temporal and spatial dynamics. Biodiversity assessments therefore have to simplify this complexity into a few facets, which are quantifiable with current knowledge and data (de Baan et al., 2012). The lack of a comprehensive approach for dealing with biodiversity impacts in LCA indicates that this area requires further consideration. Within a LCA, impacts are estimated on the basis of resources utilized and emissions released by the processes associated with the production, utilization and disposal of a product per unit of production. Many environmental impacts are commonly incorporated into a LCA, such as carbon emissions, resource depletion and eco-toxicity, but to date biodiversity has rarely been considered. Yet there is an urgent need to include biodiversity to overcome perverse outcomes that may arise when focusing on single resources, such as carbon (Penman et al., 2010).

1.5.5. Interpretation

Interpretation of LCIA (ISO, 2006b) consists of identifying significant issues, evaluation, sensitivity evaluation, assessment of data quality (Yan et al., 2010). Conclusions and recommendations are formulated to improve the production system (Crosson et al., 2011).

1.5.6. Attributional and consequential LCA

When performing an LCA, in most cases, multifunctional processes are included in the analyzed system. Choices of how to handle co-products, therefore, are inevitably connected with performing an LCA. The distinction between ALCA and CLCA was developed in the process of resolving the methodological debates over allocation problems and the choice of data (Thomassen et al., 2008b).
Consequential LCA (CLCA) and attributional LCA (ALCA) are two approaches aim to answer different questions, and failure to distinguish them can result in the wrong method being applied, a mixture of the two approaches within a single assessment, or misinterpretation of results (Brander et al., 2008).

- Attributional LCA is defined by its focus on describing the environmentally relevant physical flows to and from a life cycle and its subsystems (Finnveden et al., 2009) but does not consider indirect effects arising from changes in the output of a product. ALCA generally provides information on the average unit of product and is useful for consumption-based carbon accounting (Brander et al., 2008).
- Consequential LCA is defined by its aim to describe how environmentally relevant flows will change in response to possible decisions (Curran et al., 2005). Consequential LCA (CLCA) provides information about the consequences of changes in the level of output (and consumption and disposal) of a product, including effects both inside and outside the life cycle of the product (Brander et al., 2008).

When an attributional LCA is performed the aim of the study should be to answer to the question: "What are the total emissions from the processes and material flows used during the life cycle (production, consumption and disposal) of a product, at the current level of output?". In this case ALCA is useful for comparing the emissions from the processes used to produce (and use and dispose of) different products. It is also valuable for identifying opportunities for reducing emissions within the life cycle or supply chain, through improvements in processing efficiency or new technologies (Brander et al., 2008).
When an consequential LCA is performed the aim of the study should be to answer to the question: "What is the change (either positive or negative) in total emissions which results from a marginal change in the level of output (and consumption and disposal) of a product?". CLCA is the appropriate method for quantifying the total change emissions from a change in the level of output of a product as it takes into account both direct and indirect effects, and may therefore be of greater relevance to policy makers than ALCA (Brander et al., 2008).
A strong connection exists between the choice of ALCA and CLCA and the choice of how to handle co-products. Within ALCA, avoiding allocation by using system expansion to handle co-products is optional, while co-product allocation is most frequently used (Thomassen et al., 2008b). Avoiding allocation by system expansion, however, is the only way to deal with co-products within CLCA, as it reflects the consequences of a change in production (Weidema 2003).
In recent study Thomassen et al. (2008b) provided the findings from an ALCA and CLCA for milk production in the Netherlands. The results are shown in Table 5 below:

Tab. 5 - Results from ALCA and CLCA for Dutch milk production (Thomassen et al., 2008b).

ALCA - mass allocation (gCO_2e/1 kg milk)	ALCA - economic allocation (gCO_2e/1 kg milk)	CLCA (gCO_2e/1 kg milk)
1,560	1,610	901

The results from the CLCA for milk production were significantly lower than the results from the ALCA as the consequential method took into account emissions that would be avoided by meat from dairy cows substituting beef and pork production (which is highly carbon intensive).
Table 6 provides an overview of the main characteristics of ALCA and CLCA (Guinée et al. 2002; Weidema 2003).

Tab. 6 - Overview of main characteristics of ALCA and CLCA (Guinée et al. 2002; Weidema 2003) complemented with the comparison outcomes (from (Thomassen et al., 2008b).

	ALCA	CLCA
Synonym	Status quo	Change-oriented
Type of questions answered	Accounting	Assessing consequences of changes
Data	Average historical	Marginal future
Knowledge required	Physical mechanisms	Physical and market mechanisms
Functional unit	Represents static situation	Represents change in volume
System boundaries	Static processes	Affected processes by change in demand
System expansion	Optional	Obligatory
Co-product allocation	Frequently used	Never used
Hotspot identification	System-dependent	System-dependent
Comprehensibility		
LCA practitioner	Difficult use of arbitrary allocation factors	Difficult inclusion of future processes
Feeding expert	Good; concentrates represent reality	Difficult to understand usage of two ingredients
Quality	Sensitive to uncertainties	Higher sensitivity to uncertainties
Data availability	Similar	Similar

1.5.7. Limits of Life Cycle Assessment and the sensitive analysis

LCA is considered as the best methodology for holistic assessment of environmental impacts of a certain activity but it has its limitations (Mattila et al., 2011). Since relatively arbitrary methodological choices has to be made (i.e. co-product handling, inclusion of land use change emissions, etc.) and a limited data availability necessitates the use of several simplifications and assumptions (Gerber et al., 2010), for that reason a comparison between results obtained by different studies should be done carefully, even if the product considered in the analysis is the same.

In LCA study a sensitive analysis is recommended in order to test the variation in the results in relation to different methodological choices and to evaluate the uncertainty and the reliability of the data adopted in the analysis. In Figure 20 and Table 7 are reported the results from a sensitive analysis performed by Flysjö et al. (2011b) and Flysjö et al. (2012) respectively. As can be seen, the final outcomes (kg CO_2 eq. kg^{-1} ECM) changed widely depending which allocation method was adopted and if emission from land use change were included:

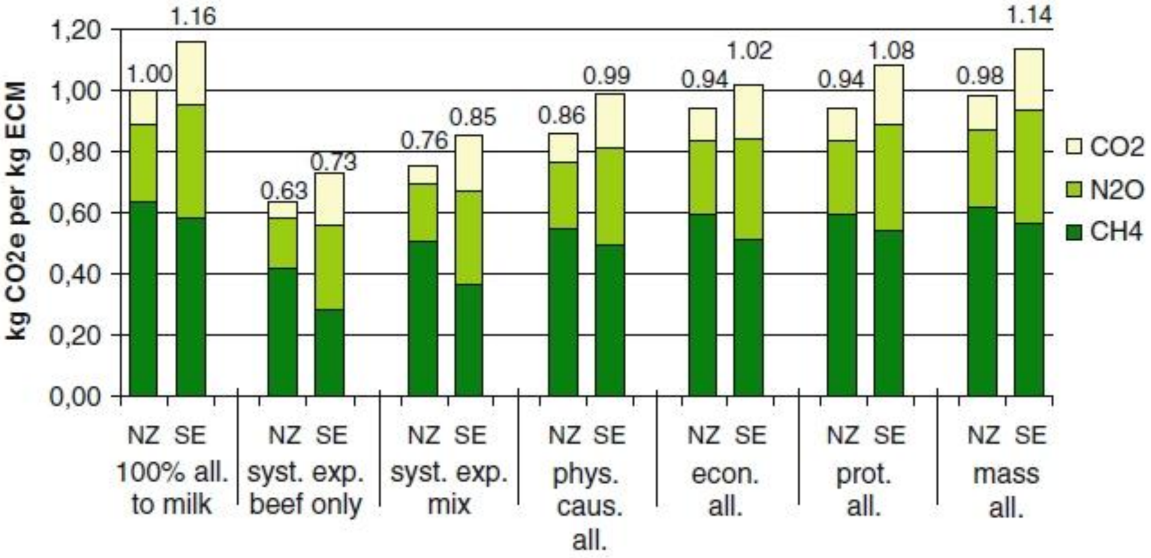


Fig. 20 - CF for one kilogram milk in NZ and SE, applying different methods of handling the co-product meat of no allocation (i.e. 100% of emissions are allocated to milk), system expansion replacing beef only, system expansion replacing a mix of meat, physical causality allocation, economic allocation, protein allocation and mass allocation (Flysjö et al., 2011b).

Tab. 7 - CF (kg CO_2-eq) for 1 kg ECM using different methods to account for GHG emissions from LUC for organic and high yielding conventional milk production in Sweden, when allocating 100% of emissions to milk (i.e. no allocation) and applying system expansion using data for EU beef (Flysjö et al., 2012).

kg CO_2e per kg ECM[a]	Organic milk		Conventional milk	
	No allocation	System expansion EU beef	No allocation	System expansion EU beef
No LUC included	1.13	0.49	1.07	0.52
LUC included for soy meal (Gerber et al., 2010)	1.23	0.56	1.42	0.85
LUC included for soy meal (Leip et al., 2010) 'medium case'	1.17	0.52	1.21	0.65
LUC included for soy meal (Leip et al., 2010) 'worst case'	1.26	0.59	1.52	0.95
LUC included for general land use (Audsley et al., 2009)	1.60	0.83	1.32	0.66
LUC included for general land use (Schmidt et al., 2011)	2.91	2.11	2.07	1.38

[a] Energy corrected milk.

In order to make future agricultural LCA studies comparable and useful for regional policy development and planning, it may be necessary for the research community to establish a recommended set of baseline categories and characterization factors (or methods to derive them locally). Choice of FU, system boundary, allocation and other assumptions give inherent uncertainty to inter-comparisons. In addition, sampling strategies should be taken into account when comparing LCAs from different countries (Yan et al., 2010).

1.5.8. Overview of results of LCAs on milk production

Within the dairy industry, from production through retail sales, the majority (80 to 95%) of global warming, eutrophication, and acidification potentials occur during the on-farm production phase (Capper et al., 2009) for that reason the majority of LCA studies on milk production were focused on the farm processes. Farm based studies indicate that there are large differences among farms in animal productivity and environmental impacts. These differences are often related to management skills of farmers, technologies applied and/or environmental conditions. Results of these farm scale studies provide insights to policy makers for possible incentives to further improve animal productivity and reduce emissions of specific farms. While comparisons at a

regional or country level are not available, results of such a study would provide information on differences in emissions among regions and could aid in identification of management practices that lower emissions (Lesschen et al., 2012).
The following tables report an overview of the LCA studies on milk production performed by several international working group, the first table (Table 8) provides data from study focused on global warming potential only, while in the second table (Table 9) are reported data of “full LCA” studies where several impact categories were considered.

Tab. 8 - Results of "cradle-to-farm gate" Carbon Footprint of milk production of different countries.

REFERENCE	Year	Country	Allocation to milk	Functional Unit	GWP (100 years-time horizon) FU^{-1}
Casey and Holden	2005a	Ireland	no	1 kg ECM	1.50 kg CO_2-eq.
			economic (85% to milk)		1.30 kg CO_2-eq.
			mass (96.6% to milk)		1.45 kg CO_2-eq.
Casey and Holden	2005b	Ireland		1 kg ECM	*REPS farm A*: 1.19 kg CO_2-eq. (6367 kg CO_2-eq. ha^{-1})
					REPS farm B: 1.51 kg CO_2-eq. (5918 kg CO_2-eq. ha^{-1})
					REPS farm C: 0.92 kg CO_2-eq. (6543 kg CO_2-eq. ha^{-1})
					REPS farm D: 1.18 kg CO_2-eq. (6145 kg CO_2-eq. ha^{-1})
					ConventionalE: 0.98 kg CO_2-eq. (8298 kg CO_2-eq. ha^{-1})
					ConventionalF: 1.28 kg CO_2-eq. (6265 kg CO_2-eq. ha^{-1})
					ConventionalG: 0.99 kg CO_2-eq. (7144 kg CO_2-eq. ha^{-1})
					ConventionalH: 1.09 kg CO_2-eq. (7529 kg CO_2-eq. ha^{-1})
					ConventionalI: 1.19 kg CO_2-eq. (7409 kg CO_2-eq. ha^{-1})
					ConventionalJ: 0.95 kg CO_2-eq. (7034 kg CO_2-eq. ha^{-1})
Capper et al.	2009	USA		1 kg milk	*year 1944*: 13.5 kg CO_2-eq.
					year 2007: 27.8 kg CO_2-eq.
				1cow	*year 1944*: 3.66 kg CO_2-eq.
					year 2007: 1.35 kg CO_2-eq.
Cederberg et al.	2009	Sweden	physical (85% to milk)	1 kg ECM	*year 1990*: 1.27 kg CO_2-eq.
					year 2005: 1.02 kg CO_2-eq.
Kassow et al.	2009	Germany		1 cow	*Conventional stable, silage, slurry*: 5.36 Mg CO_2-eq.
					Conventional pasture, silage, slurry: 5.62 Mg CO_2-eq.
					Conventional pasture, straw bed: 5.42 Mg CO_2-eq.
					Organic pasture, straw bed: 4.94 Mg CO_2-eq.
Rotz et al.	2010	USA (Pensylvania)	Economic (90-95% to milk)	1 kg ECM	*60 cow confined*: 0.69 kg CO_2-eq.
					60 cow grazing: 0.62 kg CO_2-eq.
					500 cow confined: 0.53 kg CO_2-eq.
					2000 cow drylot: 0.46 kg CO_2-eq.
		USA (California)			*500 cow confined*: 0.57 kg CO_2-eq.
					2000 cow drylot: 0.47 kg CO_2-eq.
Henriksson et al.	2011	Sweden	no	1 kg ECM	1.13 (0.1) kg CO2-eq.
Kristensen et al.	2011	Denmark	No	1 kg ECM	*Conventional*: 1.20 kg CO_2-eq.
					Organic: 1.27 kg CO_2-eq.
			Model A		*Conventional*: 1.03 kg CO_2-eq.
					Organic: 1.06 kg CO_2-eq.
			Protein mass		*Conventional*: 0.99 kg CO_2-eq.
					Organic: 1.02 kg CO_2-eq.
			Biological		*Conventional*: 0.91 kg CO_2-eq.
					Organic: 0.90 kg CO_2-eq.
			Economic		*Conventional*: 1.06 kg CO_2-eq.
					Organic: 1.10 kg CO_2-eq.
			System Expansion		*Conventional*: 0.94 kg CO_2-eq.
					Organic: 0.96 kg CO_2-eq.

Tab. 8 - Follows.

REFERENCE	Year	Country	Allocation to milk	Functional Unit	GWP (100 years-time horizon) FU^{-1}
O'Brien et al.	2011	Ireland	Biological	1 kg milk	*High grass allowance*: from 0.983 to 1.001 kg CO_2-eq.
					High concentrate: from 0.973 kg to 1.003 CO_2-eq.
					High stocking rate: from 0.942 to 1.017 kg CO_2-eq.
				1 kg milk solids	*High grass allowance*: from 12.62 to 13.28 kg CO_2-eq.
					High concentrate: from 12.28 kg to 13.41 CO_2-eq.
					High stocking rate: from 12.51 to 12.68 kg CO_2-eq.
				1 ha total area	*High grass allowance*: from 14.10 to 14.26 t CO_2-eq.
					High concentrate: from 14.97 kg to 15.18 t CO_2-eq.
					High stocking rate: from 16.75 to 17.24 kg CO_2-eq.
Zehetmeier et al.	2011	Germany	No	1 kg milk	*DC 6000*: 1.35 kg CO_2-eq.
					0.75 DC 8000: 1.13 kg CO_2-eq.
					0.6 DC 10 000: 0.98 kg CO_2-eq.
			Economic		*DC 6000*: 1.06 kg CO_2-eq.
					0.75 DC 8000: 0.93 kg CO_2-eq.
					0.6 DC 10 000: 0.89 kg CO_2-eq.
Flysjö et al.	2011a	Sweden	No	1 kg ECM	from 1.16 to 1.34 kg CO_2-eq.
		New Zealand			from 1.00 to 1.15 kg CO_2-eq.
Flysjö et al.	2011b	Sweden	No	1 kg ECM	1.16 kg CO_2-eq.
			System expansion beef only		0.73 kg CO_2-eq.
			System expansion mix		0.85 kg CO_2-eq.
			Physical causality (85% to milk)		0.99 kg CO_2-eq.
			Economic (88% to milk)		1.02 kg CO_2-eq.
			Protein (93% to milk)		1.08 kg CO_2-eq.
			Mass (98% to milk)		1.14 kg CO_2-eq.
		New Zealand	No		1.00 kg CO_2-eq.
			System expansion beef only		0.63 kg CO_2-eq.
			System expansion mix		0.76 kg CO_2-eq.
			Physical causality (86% to milk)		0.86 kg CO_2-eq.
			Economic (92% to milk)		0.94 kg CO_2-eq.
			Protein (94% to milk)		0.94 kg CO_2-eq.
			Mass (98% to milk)		0.98 kg CO_2-eq.
Vellinga et al.	2011	The Netherlands		1 kg milk	1075 g CO2-eq.
Flysjö et al.	2012	Sweden	No	1 kg ECM	<u>*No LUC included:*</u>
					Oragnic: 1.13 kg CO_2-eq.
					Conventional: 1.07 kg CO_2-eq.
					<u>*LUC included for soy meal:*</u>
					Oragnic: from 1.17 to 1.26 kg $_{CO2}$-eq.
					Conventional: from 1.21 to 1.52 kg CO_2-eq.
					<u>*LUC included for general land use:*</u>
					Oragnic: 1.60 and 2.91 kg CO_2-eq.
					Conventional: 2.91 and 2.07 kg CO_2-eq.
			System expansion		<u>*No LUC included:*</u>
					Oragnic: 0.49 kg CO_2-eq.
					Conventional: 0.52 kg CO_2-eq.
					<u>*LUC included for soy meal:*</u>
					Oragnic: from 0.52 to 0.59 kg CO_2-eq.
					Conventional: from 0.65 to 0.95 kg CO_2-eq.
					<u>*LUC included for general land use:*</u>
					Oragnic: 0.83 and 2.11 kg CO_2-eq.
					Conventional: 0.66 and 1.38 kg CO_2-eq.

Tab. 8 - Follows.

REFERENCE	Year	Country	Allocation to milk	Functional Unit	GWP (100 years-time horizon) FU^{-1}
Belflower et al.	2012	USA (Georgia)		1 kg ECM	<u>*Including biogenic sources and sinks*</u> *Pasture-based*: 0.58 kg CO_2-eq. 0.49 kg CO_2-eq. includig carbon sequestration *Confined*: 0.56 kg CO_2-eq. <u>*Excluding biogenic sources and sinks*</u> *Pasture-based*: 0.88 kg CO_2-eq. 0.79 kg CO_2-eq. includig carbon sequestration *Confined*: 0.87 kg CO_2-eq.
D. O'Brien et al.	2012b	Ireland	Physiological (from 88 to 92% to milk)	1 Livestock Unit	*Grass system*: from 3169 to 6999 kg CO_2-eq. *Confinement system*: from 7517 to 9599 kg CO_2-eq.
				1 ton milk fat plus protein	*Grass system*: from 6981 to 14924 kg CO_2-eq. *Confinement system*: from 14024 to 17497 kg CO_2-eq.
				1 ton FPCM	*Grass system*: from 521 to 1113 kg CO_2-eq. *Confinement system*: from 1034 to 1290 kg CO_2-eq.
				1 ha total area	*Grass system*: from 7154 to 15292 kg CO_2-eq. *Confinement system*: from 11107 to 13774 kg CO_2-eq.
Mc Geough et al.	2012	Eastern Canada	No	1 kg FPCM	0.92 kg of CO_2-eq.
			Economic (91% to milk)		0.84 kg of CO_2-eq.
			Dairy vs. beef animal (97% to milk)		0.90 kg of CO_2-eq.
			Physical (IDF default) (86% to milk)		0.79 kg of CO_2-eq.
			Physical (IDF specific) (73% to milk)		0.67 kg of CO_2-eq.

Tab. 9 - Results of "cradle-to-farm gate full LCA" on milk production of different countries.

REFERENCE	Year	Country	Allocation to milk	Functional Unit	GWP (100 years time horizon) FU^{-1}	AP FU^{-1}	EP FU^{-1}	Energy Use FU^{-1}	Land Use FU^{-1}
Cederberg and Mattsson	2000	Sweden	biological (85% to milk)	1000 kg ECM	*Organic*: 900 kg CO_2-eq.	15.81 kg SO_2-eq.	66 kg NO_3-eq.	3550 MJ-eq.	1925 m^2
					Conventional: 1100 kg CO_2-eq.	17.98 kg SO_2-eq.	58 kg NO_3-eq.	2511 MJ-eq.	3464 m^2
				1 ha total area	*Organic*:	52 kg SO_2-eq.	218 kg NO_3-eq.		
					Conventional:	131 kg SO_2-eq.	433 kg NO_3-eq.		
Iepema and Pijnenburg	2001	The Netherlands	economic (86% to milk)	1 kg FPCM	*Conventional*: 888 g CO_2-eq.	10 g SO_2-eq.	69 g NO_3-eq.	3.7 MJ	
					Environment-friendly: 689 g CO_2-eq.	6 g SO_2-eq.	20 g NO_3-eq.	2.4 MJ	
					Organic: 922 g CO_2-eq.	10 g SO_2-eq.	34 g NO_3-eq.	3.9 MJ	
				1 ha total area	*Conventional*:	116 g SO_2-eq.	820 g NO_3-eq.		
					Environment-friendly:	82 g SO_2-eq.	271 g NO_3-eq.		
					Organic:	115 g SO_2-eq.	396 g NO_3-eq.		
Haas et al.	2001	Germany	no	1 ton milk	*Intensive*: 1.3 (1.1–1.7) t CO_2-eq.	19 kg SO_2-eq.	7.5 kg PO_4^{3-}-eq.	2.7 (1.6–3.9) GJ-eq.	
					Extensive: 1.0 (0.9–1.2) t CO_2-eq.	17 kg SO_2-eq.	4.5 kg PO_4^{3-}-eq.	1.3 (1.0–1.6) GJ-eq.	
					Organic: 1.3 (1.2–1.4) t CO_2-eq.	22 kg SO_2-eq.	2.8 kg PO_4^{3-}-eq.	1.2 (0.8–1.8) GJ-eq.	
				1 ha farmed grassland	*Intensive*: 9.4 (7.5–11.2) t CO_2-eq.	136 (119–145) kg SO_2-eq.	54.2 (17.8–90.1)kg PO_4^{3-}-eq.	19.1 (10.4–28.7) GJ-eq.	
					Extensive: 7.0 (5.7–8.0) t CO_2-eq.	119 (96–143) kg SO_2-eq.	31.2 (0.6–48.6) kg PO_4^{3-}-eq.	8.7 (5.5–12.2) GJ-eq.	
					Organic: 6.3 (5.6–7.3) t CO_2-eq.	107 (94–118) kg SO_2-eq.	13.5 (7.4–19.0) kg PO_4^{3-}-eq.	5.9 (3.8–10.6) GJ-eq.	
Cederber and Flysjö	2004	Sweden	economic (90% to milk)	1 kg ECM	*Conventional H*: 896.22 (38.38) g CO_2-eq.	NH_3: 4.65g (0.34); NOx: 1.27g (0.06) SO_2: 0.60g (0.036)	NO_3: 17.47g (3.03); NH_3: 4.65g (0.34) NOx: 1.27g (0.06)	2.59 (0.12) MJ	1.54 (0.18) m^2
					Conventional M: 1037.31 (41.74) g CO_2-eq.	NH_3: 4.44g (0.37); NOx: 1.30g (0.06) SO_2: 0.58g (0.039)	NO_3: 21.80g (2.96); NH_3: 4.44g (0.37) NOx: 1.30g (0.06)	2.73 (0.13) MJ	1.92 (0.19) m^2
					Organic: 938.49 (48.2) g CO_2-eq.	NH_3: 5.63g (0.43); NOx: 1.07g (0.09) SO_2: 0.30g (0.045)	NO_3: 27.57g (3.79); NH_3: 5.63g (0.43) NOx: 1.07g (0.09)	2.10 (0.15) MJ	2.93 (0.22) m^2
Thomassen and de Boer	2005	The Netherlands	economic (88.8% to milk)	1 kg FPCM	*Organic*: 1.81 (0.86) kg CO_2-eq.	11.81 (2.14) g SO_2-eq.	82.14 (38.58) g NO_3-eq.	2.48 (0.91) MJ	1.60 (0.30) m^2
				1 ha total area	*Organic*:	161.12 (114.05) kg SO_2-eq.	1127.04 (854.28) kg NO_3-eq.		
				1 ha farm area	*Organic*:	74.15 (18.74) on-farm kg SO_2-eq.	341.69 (277.92) on-farm kg NO_3-eq.		
Thomassen et al.	2008a	The Netherlands	economic (90% to milk)	1 kg FPCM	*Organic:* 1.5 (0.3) kg CO_2-eq.	10.8 (1.9) g SO_2-eq.	0.07 (0.03) kg NO_3-eq.	3.1 (0.88) MJ	1.8 (0.4) m^2
			economic (91% to milk)		*Conventional*: 1.4 (0.1) kg CO_2-eq.	9.5 (0.8) g SO_2-eq.	0.11 (0.01) kg NO_3-eq.	5.0 (0.6) MJ	1.3 (0.1) m^2

Tab. 9 - Follows

REFERENCE	Year	Country	Allocation to milk	Functional Unit	GWP (100 years time horizon) FU^{-1}	AP FU^{-1}	EP FU^{-1}	Energy Use FU^{-1}	Land Use FU^{-1}
Basset-Mens et al.	2009	New Zealand	biological (85% to milk)	1 kg of milk	*Average NZ farm*: 0.933 kg CO_2-eq.	0.00812 kg SO_2-eq.	0.00293 kg PO_4^{3-}-eq.	1.51 MJ LHV	1.15 m^2
					Low Input farm: 0.646 kg CO_2-eq.	0.00385 kg SO_2-eq.	0.00159 kg PO_4^{3-}-eq.	0.55 MJ LHV	0.74 m^2
					N fertilzer farm: 0.762 kg CO_2-eq.	0.00674 kg SO_2-eq.	0.0025 kg PO_4^{3-}-eq.	1.13 MJ LHV	0.8 m^2
					N fertilzer + maize silage farm: 0.754 kg CO_2-eq.	0.00578kg SO_2-eq.	0.00238 kg PO_4^{3-}-eq.	1.55 MJ LHV	0.72 m^2
				1 ha total area	*Average NZ farm*: 8136 kg CO_2-eq.	70.8 kg SO_2-eq.	25.5 kg PO_4^{3-}-eq.	13186 MJ LHV	
					Low Input farm: 8694 kg CO_2-eq.	51.8 kg SO_2-eq.	21.4 kg PO_4^{3-}-eq.	7327 MJ LHV	
					N fertilzer farm: 9553 kg CO_2.eq.	84.5 kg SO_2-eq.	31.4 kg PO_4^{3-}-eq.	14110 MJ LHV	
					N fertilzer + maize silage farm: 10453 kg CO_2-eq.	80.1 kg SO_2-eq.	33.0 kg PO_4^{3-}-eq.	21470 MJ LHV	
Thomassen et al.	2009	The Netherlands	economic	1 kg FPCM	*Conventional*: 1.36 (0.3) kg CO_2-eq.	11.2 (2.6) g SO_2-eq.	0.12 (0.04) kg NO_3-eq.	5.30 (1.3) MJ	1.28 (0.4) m^2
				1 ha total area	*Conventional*:	95 (19) kg SO_2-eq.	976 (334) kg NO_3-eq.		
van der Werf et al.	2009	France	economic	1000 kg FPCM	*Organic:* 1082 (c.v: 12%) kg CO_2-eq.	6.8 (c.v: 16%) kg SO_2-eq.	5.0 (c.v: 74%) kg PO_4^{3-}-eq.	2.6 (c.v: 34%) GJ	2085 (c.v. 16%) m^2
					Conventional: 1037 (c.v: 14%) kg CO_2-eq.	7.6 (c.v: 16%) kg SO_2-eq.	7.1 (c.v: 37%) kg PO_4^{3-}-eq.	2.8 (c.v: 16%) GJ	1374 (c.v: 18%) m^2
				1 ha total area	*Organic:* 4887 (c.v: 16%) kg CO_2-eq.	31.0 (c.v: 22%) kg SO_2-eq.	20.7 (c.v: 88%) kg PO_4^{3-}-eq.	12.1 (c.v: 30%) GJ	
					Conventional: 6271 (c.v: 17%) kg CO_2-eq.	48.1 (c.v: 16%) kg SO_2-eq.	39.8 (c.v: 35%) kg PO_4^{3-}-eq.	18.9 (c.v: 15%) GJ	
Castanheira et al.	2010	Portugal	economic (87% to milk)	1 ton milk	*Typical dairy farm*: 1021 kg CO_2-eq.	20 kg SO_2-eq.	7.1 kg PO_4^{3-}-eq.		
Müller-Lindenlauf et al.	2010	Germany	no	1 kg milk	*Organic ext. grass.*: 1172 (753–1264) g CO_2-eq.			1.20 (1.04–1.51) MJ	3.1 (2.1–4.0) m^2
					Organic int. grass.: 1036 (853–1048) g CO_2-eq.			1.52 (1.17–1.93) MJ	2.7 (1.8–2.7) m^2
					Organic ext. til.: 1082 (645–1441) g CO_2-eq.			1.32 (0.87–1.97) MJ	2.8 (1.9–4.7) m^2
					Organic int. til.: 917 (763–1033) g CO_2-eq.			1.17 (1.14–1.57) MJ	2.2 (1.6–2.3) m^2
Bartl et al.	2011	Perù	economic (63% to milk)	1 kg ECM	*Highlands*: 13.78 kg CO_2-eq. (20 yr time horizon)	14.13 g SO_2-eq.	15.47 g PO_4^{3-}-eq.	0.2 MJ	23.11 m^2
			economic (96% to milk)		*Coast*: 3.18 kg CO_2-eq.(20 yr time horizon)	7.55 g SO_2-eq.	4.84 g PO_4^{3-}-eq.	4719.7 MJ	1.03 m^2
			economic (37% to animal)	1 animal (average L.W)	*Highlands*: 2846 kg CO_2-eq. (20 yr time horizon)	2918 g SO_2-eq.	3195 g PO_4^{3-}-eq.		
			economic (3% to animal)		*Coast*: 8066 kg CO_2-eq. (20 yr time horizon)	19174 g SO_2-eq.	12296 g PO_4^{3-}-eq.		
				ha farm area	*Highlands*:	4442 g SO_2-eq.	4864 g PO_4^{3-}-eq.		
					Coast:	27482 g SO_2-eq.	17625 g PO_4^{3-}-eq.		

Tab. 9 - Follows

REFERENCE	Year	Country	Allocation to milk	Functional Unit	GWP (100 years time horizon) FU^{-1}	AP FU^{-1}	EP FU^{-1}	Energy Use FU^{-1}	Land Use FU^{-1}
Oudshoorn et al.	2011	Denmark		1 kg ECM	*Organic BAU.*: 1.32 kg CO_2-eq.		N surplus: 12 g (N surplus area^{-1}: 117 kg ha^{-1})	3.22 MJ	
					Organic ANW.: 1.48 kg CO_2-eq.		N surplus: 16 g (N surplus area^{-1}: 116 kg ha^{-1})	3.33 MJ	
					Organic ENV: 1.25 kg CO_2-eq.		N surplus: 15 g (N surplus area^{-1}: 80 kg ha^{-1})	2.85 MJ	
Fantin et al.	2012	Italy	no	1 l of milk	*Conventional*: 1.1 kg CO_2-eq.	0.02 kg SO_2-eq.	0.008 kg PO_4^{3-}-eq.		
O'Brien et al.	2012a	Ireland	biological (88% to milk)	1 ton FPCM	*Grass-based*: 874.3 kg CO_2-eq.	6.9 kg SO_2-eq.	3.4 kg PO_4^{3-}-eq.	2.3 GJ	727.9 m^2
			biological (91% to milk)		*Confinement*: 1027.4 kg CO_2-eq.	11.9 kg SO_2-eq.	4.6 kg PO_4^{3-}-eq.	3.9 GJ	933.3 m^2
				1 ton milk solids	*Grass-based*: 11721.8 kg CO_2-eq.	93.1 kg SO_2-eq.	44.9 kg PO_4^{3-}-eq.	30.4 GJ	9759.3 m^2
					Confinement: 13938.9 kg CO_2-eq.	161.9 kg SO_2-eq.	62.7 kg PO_4^{3-}-eq.	53.2 GJ	12661.7 m^2
				1 ha farm area	*Grass-based*: 13529 kg CO_2-eq.	107.5 kg SO_2-eq.	51.8 kg PO_4^{3-}-eq.	35.1 GJ	
					Confinement: 37499 kg CO_2-eq.	435.4 kg SO_2-eq.	168.7 kg PO_4^{3-}-eq.	143 GJ	
				1 ha total area	*Grass-based*: 12011 kg CO_2-eq.	95.4 kg SO_2-eq.	46.0 kg PO_4^{3-}-eq.	31.2 GJ	
					Confinement: 10907 kg CO_2-eq.	126.7 kg SO_2-eq.	49.1 kg PO_4^{3-}-eq.	41.6 GJ	

1.6. MITIGATION STRATEGIES AT THE FARM LEVEL

Environmental considerations are increasingly being given higher priority particularly in agricultural issues. Food production has an environmental impact, so as global populations continue to increase, it is critical that sufficiently high-quality food be produced from a finite resource supply and that effects upon the environment be minimized (Meneses et al., 2012). In the last decades several research groups (not only in Europe but also in Australia and North America) focused to assess the environmental impact of agricultural products throughout their life cycle with the help of LCA (Life Cycle Assessment). The strength of a "life cycle thinking" is, over the estimation of a total environmental burden of a production process, to identify how the different life cycle stages contribute to the environmental impact so that a more sustainable production can be developed.

GHG emissions from raw milk production at farm level have a dominating influence (70-90%) on the environmental impact of the carbon footprint of dairy products (Flysjö, 2011) but also regarding the other impact categories (for instance acidification and eutrophication potential) the weight of the primary production is very high (Fantin et al., 2012; Table 10).

Tab. 10 – Impact assessment results for 1 l of high quality milk and percentage contribution of each life cycle phase (Fantin et al., 2012).

Impact categories	Units	Total	From cradle to farm gate	Transport to dairies	Dairies	Transport to distribution centres
Global warming	kg CO_2 eq.	1.3	82%	1%	14%	3%
Ozone layer depletion	kg CFC-11 eq.	6.2E-08	59%	3%	29%	9%
Photochemical oxidation	kg C_2H_4 eq.	2.2E-04	82%	1%	14%	3%
Acidification	kg SO_2 eq.	2.1E-02	96%	<1%	2%	1%
Eutrophication	kg PO_4^{3-} eq.	8.0E-03	97%	<1%	2%	<1%

For that reason identify and evaluate the real effectiveness of some production systems, of the technical strategies and managing option in order to mitigate the environmental burden of the farm stage is currently an important topic and it will be probably the future challenge for a more environmentally friendly way to produce.

Many recent LCA studies investigated the environmental impact of different farming systems, for instance organic vs. conventional (Cederberg and Mattson, 2000; de Boer, 2003; Thomassen et al., 2008a and Kristensen et al., 2011) or confinement vs. grass-based (O'Brien et al., 2012a; Belflower et al., 2012) but right now there is not a shared consensus of which is the best system especially when the impact is estimated on the product base. Other work evaluated how changes in farming management can affect the environmental performance of milk production.

In the following section will be reported some of the strategies that could be adopted at the farm level in order to make the milk production more sustainable. All of these mitigation options were analyzed by several research groups and not all of them showed similar effectiveness in term of mitigation. Anyway is not easy to evaluate the real effect of mitigation strategy especially in a global perspective when many factors are involved and strictly connected each other. In general the impacts of improved genetics, fertility and health all contribute to reducing the number of animals required to meet a steady demand for animal products, while the issues of feed, manure and grazing management are rather more complex (Gill et al., 2010).

1.6.1. Improving animal efficiency and productivity

The potential mitigation which is still to be captured from improved productivity is obviously dependent on the basal level of productivity and is greater in developing countries (Gill et al., 2010), however it is widely recognized that improving animal efficiency (in term of feed conversion rate) has a positive effect on environmental impact because the animals can produce the same amount of product with lower feed ingestion (less feed is required, that means less emissions)

(Hermansen and Kristensen, 2010; de Boer et al., 2011; Opio et al., 2012). When an higher milk yield per head is achieved less cows are needed to produce the same amount of milk (Capper et al., 2008) moreover cows that produce more milk reduce the proportion of total consumed feedstuffs going toward maintenance energy costs (Place and Mitleohner, 2010). The higher animal efficiency and production is the results of both genetic, managerial and technical choices. Past selection for production traits such as growth rate, milk production, fertility and efficiency of feed conversion has resulted in decreases in GHG production per unit of livestock product of about 1% per annum (Gill et al., 2010). However in developed countries, where animal productivity is already high, genetic selection for growth rate or annual milk production per cow might negatively affect animal health or fertility or the social acceptance of animal production (de Boer et al., 2011). If a breeding strategy, aimed at improving lactational performance, resulted in impaired fertility and, consequently longer calving intervals and higher culling rate, overall emissions may increase (Crosson et al., 2011). Moreover, in an evaluation of the global impact of the livestock sector, it should be taken into account that less dairy cows means less surplus calves. Considering that in Europe approximately 50% of the beef production is derived from co-products from the dairy sector (Cederberg and Stadig, 2003) to supply the lack of surplus calves more beef cow are bred and the environmental impact of this process offsets the mitigating effect of improving productivity in the dairy sector (Zehetmeier et al., 2012). Several authors investigated the effect of increasing in milk productivity on GWP of different farming systems. Rotz et al. (2010) simulated the benefits of improved animal genetics and feeding management on milk production and farm environmental performances: milk production was maximized for the given feeding strategy, feed intake increased to meet the nutrient requirements of the larger, higher producing animals and this increased CH_4 and CO_2 emissions. More manure was also produced, which increased manure storage emissions. With greater feed use, cropland provided a greater sink of CO_2, but fuel combustion and secondary emissions both increased. Overall the net GHG emission increased 6%, but the greater milk production reduced the carbon footprint by 8%. Moreover Rotz et al. (2010) showed that, when recombinant bST (bovine somatotropine) was included for even greater milk production, feed intake increased, resulting further GHG emissions from the animals, from manure storage, from fuel combustion and secondary sources, so the net GHG emission increased another 1% (compared with the previous strategy without the use of rbST) but the carbon footprint decreased an additional 7%. O'Brien et al. (2012b) observed that increasing milk production through genetic improvement increased GHG emissions per livestock unit, however, the increase in milk production was greater than the increase in dairy systems' GHG emissions. Therefore, GHG emissions per unit of product decreased. Belflower et al. (2012) found that a 22% increase in milk production reduced the footprint of a pasture-base dairy by about 15% whether the increased production was obtained through animal management or feeding more corn. In the work of Vellinga et al. (2011) the increase of milk production per cow was not effective in reducing GHG emissions, the study indicated that in the range of 7500 to 9000 kg milk the options for increasing resource use efficiency by increasing milk production per cow are very limited. He pointed out that the focus should be on realizing an increased feed efficiency, rather than on high milk productions per sé. Eckard et al. (2010) pointed out that Improving N efficiency and reducing excess urinary N can be achieved through either breeding animals with improved N efficiency, breeding forages that use N more efficiently and have a higher energy-to-protein ratio or balancing high protein forages with high-energy supplements.

1.6.2. *Increasing fertility and reproduction*

Reproductive success is influenced by nutrition, genetics, health disorders during transition, management, and the environment. Reproductive performances greatly affect emissions per kilogram of milk. Dairy cows that have extended calving intervals because of conception failure spend more time out of peak milk when feed conversion into milk is most efficient (Place and Mitleohner, 2010). The reproductive performance and productive traits of dairy cattle are negatively related. The negative effect per kilogram of milk emissions caused by declining reproductive efficiency has likely been offset by increases in milk production per cow (Place and Mitleohner, 2010). However Garnsworthy (2004) found that both higher milk yield and improved reproductive performance (better estrus detection and conception rates) contributed to reduced CH_4 and NH_3 emissions because of the smaller lactating and replacement herd population required to meet UK production quotas. "Intensified" feeding programs for dairy heifers have been shown to lower age at first calving with no reduction or even an improvement in first lactation milk yield. Both decreasing the age at first calving and increasing first lactation yield could improve milk's life-cycle production efficiency and decrease emission per kilogram of milk (Place and Mitleohner, 2010). Sexed semen, if used selectively, can increase the rate of genetic gain in dairy cattle, allowing advantageous traits to become ubiquitous in the entire dairy cattle population (De Vries et al., 2008). Furthermore, on average, heifer calves are smaller than bull calves and cause fewer dystocias, which may allow for earlier breeding of heifers, and fewer mortalities and health problems (Weigel, 2004).

1.6.3. Reducing the replacement rate

Garnsworthy (2004) reported that replacements contributed up to 27% of the methane and 15% of the ammonia attributed to dairy cows in UK. Reducing the replacement rate means that an average cow undergoes more lactations and dry-off periods than in the reference situation. An increase of lactations per cow has the potential to reduce GHG emissions, as heifers emit greenhouse gases without producing milk (Weiske et al., 2006). Vellinga et al. (2011) showed that a reduction of the replacement rate with 5 to 9% reduced the GHG emissions by 20 to 39 g per kg milk. Increasing longevity of milking cows by breeding or management decreases number of replacements, and, therefore is stated to reduce GHG emissions of milk production in developed countries. Similarly Weiske et al. (2006) estimated that the combination of a reduction in replacement rate and selling heifers just after birth reduced GHG emissions. Increasing longevity without increasing excess progeny, by extending the lactation period in combination with increasing calving interval, might have higher potential for net GHG reduction (de Boer et al., 2011). Similarly Eckard et al. (2010) in a recent review reported that strategies such as extended lactation in dairying, where cows calve every 18 months rather than annually, reduce herd energy demand by 10.4% and thus potentially reduce on-farm CH_4 emissions by a similar amount. O'Brien et al. (2012b) observed that reducing the replacement rate had little effect on GHG emissions per unit of product because the percentage of GHG emissions allocated to milk rather than to beef was increased. Regarding the use of sexed semen it should be considered that if all animals are bred with sexed semen (or even all heifers), the replacement population for the dairy herd will increase in size. To keep the total population of dairy cattle at a level that does not create an oversupply of milk, the lactating cow cull rate must increase. In the context of environmental impact per kilogram of milk, the widespread use of sexed semen could increase emissions per kilogram of milk by shortening the total productive lifetime of dairy cows and a larger replacement herd size means more nonproductive emissions for each kilogram of milk produced (Place and Mitleohner, 2010). Weiske et al. (2006) pointed out that changing the replacement rate as a management-oriented mitigation measure is applicable to all dairy production systems without any technical efforts or additional expenditures, however, this option was only efficient if the surplus heifers were slaughtered after selling.

1.6.4. Herd health

The impact of disease on livestock productivity is highly variable between countries dependent on the incidence of endemic diseases, and between years on the incidence of infectious diseases, particularly when these are associated with the culling of animals (Gill et al., 2010). Herd-health challenges affect per unit of milk emissions by increasing mortality and losses of saleable milk and decreasing reproductive performance and milk production efficiency. Herd health is influenced by many factors, including management, nutrition, the environment, and social stressors (Place and Mitleohner, 2010). All the diseases and environmental or social stressors can decrease the production efficiency of the cow and subsequently increase the emissions of each kilogram of milk that she produces (Place and Mitleohner, 2010). For instance excessive negative energy balances during the transition period, heat stress, mastitis and lameness can decrease the production efficiency of the cow and subsequently increase the emissions of each kilogram of milk that she produces (Place and Mitleohner, 2010). Moreover Place and Mitleohner (2010) underlined that health of dairy calves depends on passive immunization from the absorption of antibodies in colostrum to provide adequate immunity during their early life stages. Failure of passive transfer of immunity leads to increased mortality and morbidity and decreased growth performance. With regard to social stress, grouping animals according to size and age and minimizing overcrowding can improve dry matter intake, consequentially improving milk production. Improving cow cooling during hot summer months and grouping animals to minimize behavioral stress has been the focus of research to improve farm profitability, but these improvements have the potential to decrease emissions per kilogram of milk as well (Place and Mitleohner, 2010).

1.6.5. Reducing enteric emissions

Research in methane was common in the 1960s when various ruminant researches tried to decrease methane production as a means of achieving increased feed conversion ratio (unit of feed in : unit of product out), since eructation of methane represent loss of energy to the animal (Gill et al., 2010). Typically, about 6–10% of the total gross energy consumed by the dairy cow is converted to CH_4 and released via the breath. Therefore, reducing enteric CH_4 production may also lead to production benefits (Eckard et al., 2010). Options to reduce enteric CH_4 emissions in ruminants focus on breeding, feeding, management and dietary supplementation and are explored mainly for developed countries (de Boer et al., 2011). Apparently significant successes in decreasing methane production have been achieved in experiments in vitro or in single animal feeding trials but these have not proved to be robust when applied to a variety of feeding regimes and some methods such as the use of ionophores are banned in the European Union (Gill et al., 2010). Diet composition can alter rumen fermentation to reduce the amount of CH_4 produced (Ellis et al., 2008) and the NH_3 emissions produced from the manure (Van de Haar and St-Pierre, 2006). The substrates used by methanogens are byproducts of structural carbohydrate fermentation; thus, high concentrate diets containing more nonstructural carbohydrates can lead to decreased CH_4 emissions (Ellis et al., 2008). Increasing the energy density of the diet (e.g. by increasing ratio of concentrates to forage) decreases methane production per unit of digestible energy ingested, moreover it also increases productivity, thereby also contributing to decreased carbon per unit of product (Gill et al., 2010). de Boer et al. (2011) underlined that potential feeding strategies are replacing grass silage by maize silage; increasing the ratio of concentrates over roughage; improving forage digestibility or quality, or dietary supplementation. Replacing grass silage by maize silage, however, not only reduces enteric CH_4 emissions but also affects the farm plan, that is, the transition of grassland into maize land (loss of soil carbon) and the type and amount of purchased concentrates. The net GHG reduction along the chain, therefore, is not self-evident. Similarly, increasing the ratio of concentrates over roughage reduces enteric CH_4

emissions but might increase CO_2 and N_2O emissions during production and transport of additional concentrates or CH_4 emissions during manure storage. A high ratio of concentrates in a cow's ration increases the risk for rumen acidosis or lameness. Furthermore, Place and Mitleohner (2010) observed that very high concentrate diets diminish the principal environmental benefit of dairy cows: their ability to convert cellulose, indigestible to humans and the Earth's most abundant organic molecule, into high-quality proteins for human consumption. Therefore, the CH_4 produced by dairy cattle cannot simply be seen as a gross energy loss and greenhouse gases source but is a necessary consequence of transforming inedible fibrous forages and byproducts (e.g., almond hulls, citrus pulp, distillers grains) into food and fiber products fit for human use. For that reason another option to reduce enteric methane production is to improve forage quality because tends to increase the voluntary intake and reduces the retention time in the rumen, promoting energetically more efficient post-ruminal digestion and reducing the proportion of dietary energy converted to CH_4, methane emissions are also commonly lower with higher proportions of forage legumes in the diet, partly because of the lower fiber content, the faster rate of passage, and in some cases, the presence of condensed tannins (CTs) (Eckard et al., 2010). In a recent study Rotz et al. (2010) estimated that maximize the use of forage in all animal diets instead concentrates had the consequence of more CH_4 produced by the animals due to more fiber in diets, moreover excreted volatile solids (VS) were also greater, creating a small increase in emissions from the manure storage. In the same work Rotz et al. (2010) found that shifting the quota of maize silage from 50 to 75% (and Lucerne from 50 to 25%) of the forage fed to the herd the consequent change in fiber and starch contents in diets reduced animal and manure storage emissions of CH_4 and CO_2. Feed N was also used more efficiently, which provided a small decrease in excreted manure N and the resulting production of N_2O from cropland. O'Brien et al. (2012b) observed that improving forage quality (in order to t achieve a higher net energy content (+0.05 unit feed lactation UFL kg^{-1} DM) reduced total GHG emissions per unit of product . The strategy reduced CH_4 emissions from enteric fermentation and manure storage, N_2O emissions from fertilizer use and manure excreted by grazing animals and off-farm emissions from concentrate and fertilizer production. Modeling different farm strategies, Vellinga et al. (2011) found that an higher maize fraction in the ration increases the digestibility of the roughage and reduced the concentrate input but the total feed intake was only slightly reduced for that reason CH_4 emissions from enteric fermentation, which are related to feed intake, were reduced to a limited extent. Moreover the lower N content of the maize, compared to the grass reduces the N intake and thus affects manure quality. This leads to changes in fertilizer use and N_2O emissions. The most important reduction is realized in CO_2 emissions, due to the fact that less energy consuming concentrates are used.

Substantial reductions in CH_4 emissions can be achieved without feeding high levels of concentrates by altering the previously mentioned nutritional factors: microbial-altering feed additives, dietary lipids, and forage processing and quality. Feed additives, such as the ionophore monensin, can change microbial processes in the rumen to potentially improve feed efficiency and reduce CH_4 emissions, however its use is not allowed in European Union. Alternatives to ionophores such as probiotics (e.g., yeast), essential oils and biologically active plant compounds (e.g., condensed tannins) have shown promise for CH_4 reductions (Place and Mitleohner, 2010). Dietary lipids, specifically unsaturated fatty acids, have the potential to act as an alternate H sink in the rumen, thereby reducing the H available to methanogens and the CH_4 produced (Ellis et al., 2008). Promising dietary supplements to reduce enteric CH_4 emissions include linseed, clover and ionophores. Except for ionophores, which are prohibited as a dietary supplement in the European Union, evidence for GHG reduction of these supplements generally is based on short-term in vivo or in vitro experiments (de Boer et al., 2011).

1.6.6. *Improving manure management*

In a recent review de Boer et al. (2011) observed that manure management options focus mainly on reduction of N_2O and CH_4 emissions by changes in livestock buildings, manure storage facilities, manure treatment and grazing management: reduction of N_2O can be achieved by adopting slurry-based instead of straw or deep-litter systems, on the other hand, in absence of bedding material, slurry remains in a predominantly anaerobic state with little opportunity for nitrification, and hence N_2O emissions but the opportunity for CH_4 formation in this situation, however, might increase. O'Brien et al. (2012a) estimated that storing manure in solid rather than liquid systems caused a greater reduction in environmental impacts for a confinement farming system relative to the grass-based system, because of the longer housing period. de Boer et al. (2011) also pointed out that a regular removal of manure from buildings and storage facilities reduces CH_4 formation, as does minimizing storage of slurry in summer or cooling slurry, to cool manure, however, energy can be needed (which can be produced sustainably), whereas minimizing storage of manure is possible only when there are direct opportunities for manure application. Weiske et al. (2006) estimated that daily removal of manure may reduce emissions at the whole farm level, however, it has to be taken into account that preventing losses of NH_3 from housing and storage results in a higher nutrient concentration in the manure. In the same study Weiske et al. (2006) found that the use of scrapers for reducing NH_3 emissions increased GHG emissions at the farm level because, although scraping reduced indirect N_2O emissions derived from NH_3 volatilization, the additional GHG emissions predicted during prolonged outside storage and after field application were much higher (increase in emissions from agricultural soils due to an higher emission factor for nitrate leaching than for NH_3 volatilization).

Decreasing grazing time, decreases the amount of N excreted at urine spots, and, hence, N_2O emissions and nitrate leaching (de Boer et al., 2011), but increase NH_3 and CH_4 emissions from storage because of increased time spend inside the house where manure is stored in the slurry pit (de Boer et al., 2011). Anaerobic digestion of, for example, pig manure reduces the environmental impact of manure management by reducing storage emissions and substituting fossil fuel, but current efficiency of bio-gas production from manure only is low (de Boer et al., 2011). Co-substrates (i.e. maize silage, glycerine, food waste, or enzymes) are generally added to increase efficiency in bio-gas production but they increases the amount of nutrients in the remaining digested product, and, therefore, requires a larger land area for application (de Boer et al., 2011). Addition of these co-substrates, however, can increase acidification, eutrophication and land use because alternative products need to be produced to substitute these co-substrates, as they are no longer available for their original use (e.g. feed ingredient) (de Boer et al., 2011). Another aspect of biogas installation is, for example, that would favor high animal numbers and indoor production systems that could have other detrimental implications (Bellarby et al., 2013). Nonetheless O'Brien et al. (2012b), observed that covering the manure store and flaring CH_4 reduced on-farm and total GHG emissions per unit of product and area, this kind of manure management eliminated N_2O emissions from slurry storage and significantly reduced CH_4 emissions. Similar results were found by Rotz et al. (2010) who estimated a net reduction of 39% in the net GHG emission of milk production when an enclosed manure storage was used with a flare to burn the escaping biogas. This management change almost eliminated CH_4 emission from the storage, but CO_2 emission increased. With the enclosed storage, a crust would not form, which eliminated N_2O formation and emission from the storage. A substantial reduction in the carbon footprint of milk due to the use of the manure storage cover and the burning of biogas was observed also by Belflower et al. (2012). Weiske et al. (2006) pointed out that the extent of the reduction of greenhouse gases emissions from biogas production by anaerobic digestion and flaring depends on how much of the thermal energy produced is used to substitute fossil fuels.

Vellinga et al. (2011) estimated that an improved manure utilization by increasing the manure storage capacity and applying the manure earlier in the growing season slightly reduced the emissions. Improved manure application techniques is particularly important to control greenhouse gases emissions because without an improved application at the end-of-pipe of dairy production manure handling, much of the benefit of preliminary mitigation measures during animal housing and manure storage may be lost (Weiske et al., 2006). For example, the injection or incorporation of effluent into the soil can increase direct N_2O emission but reduce ammonia (NH_3) volatilization, resulting in lower indirect N_2O emissions. Effluent injection is also likely to increase the overall efficiency of the use of effluent N and could thus reduce the N fertilizer requirement and the associated N_2O emissions (Eckard et al., 2010). The use of trail hose or injection may reduce odor emissions, as well as NH_3 volatilization, moreover, an optimized application technique leads to less crop coverage and forage contamination with manure and, therefore, to an improved growth and quality of crops (Weiske et al., 2006).

1.6.7. Land use and carbon sequestration

Smith et al. (2008) estimated the potential of a range of land management practices to mitigate GHG emissions, identifying restoration of organic soils, management of cropland and grassland as having particularly high potential, though there are issues associated with permanence and saturation of the carbon sink.

Measures that increase carbon input into the soil include: use of manure on crop instead of grassland; improved rotations with higher carbon input to soil (catch crop); increased crop yield and hence the related crop residues, for example, by better plant breeding, crop husbandry, irrigation or fertilization and conversion from arable land to grassland or grazing management (de Boer et al., 2011). Crosson et al. (2011) reported that, as well as being large carbon sinks, permanent grassland soils can have an important role in sequestering carbon, particularly where improved grazing strategies have been adopted. Soussana et al. (2010) suggested that grasslands range from sinks to sources depending on climate, management and site characteristics such as soil type. Even though there is the potential of fully compensating for beef or dairy emissions at the farm level, there is the risk that accumulated carbon can be lost in events such as an unusually dry summer. At present, the uncertainty about the potential of most of measures to store additional carbon, the rate of accumulation of soil carbon, and the permanence of this carbon sink is high. In addition, their life cycle budget regarding all greenhouse gases is unknown (de Boer et al., 2011). Moreover, increased carbon sequestration by a management practice may increase other emissions and, as such, decrease or even negate the sequestered CO_2 in the soil (Bellarby et al., 2013). In a recent study Belflower et al. (2012) observed that when the potential sequestration of carbon in the soil of the perennial grassland was considered, the carbon footprint of milk from the pasture-based dairy was 12% less than that of the confinement dairy. Similar results were found by Rotz et al. (2010) and O'Brien et al. (2012a). Rotz et al., (2010) estimated a reduction by 10 to 22% of milk carbon footprint as the effect of transitioning a farm with confinement feeding of rotated crops to that including permanent pasture, but during this transition period of up to 50 years, the carbon footprint would gradually increase. O'Brien et al. (2012a, 2012b) found that GHG emission of two dairy systems was reduced by including grassland carbon sequestration in the model estimation, but carbon sequestration had no effect on the remaining environmental impacts. In general carbon sequestration by soil under pasture would slow over time as the soil reaches a new level of equilibrium, and this benefit would diminish Belflower et al. (2012).

1.6.8. Improving crop production and field operation

Emissions of CO_2 and N_2O from production of feed ingredients can be reduced by selection of crops with a higher yield (or lower N demand per unit output) (de Boer et al., 2011). Over an higher yield, plant breeding can potentially improve digestibility as well as reduce CH_4, in fact improving forage quality can both improve animal performance and reduce CH_4 production, but it can also improve efficiency by reducing CH_4 emissions per unit of animal product (Eckard et al., 2010). In the study of Vellinga et al. (2011) was shown that when more feed is produced at the farm, more emissions are taken into account and emissions on farm scale increase, however, less external land is needed to produce the same amount of feed and emissions are prevented. Thus, on a regional scale emissions will reduce by this improved resource use efficiency. Rotz et al. (2010) observed that, if the farm was modified to maximize the use of forage in all animal diets, harvesting of the additional forage required more machinery operations and fuel compared with the grain feed replaced, which increased combustion and secondary emissions and this led to an 18% increase in net GHG emission and C footprint. However if lucerne was partially substitute by maize silage the crop operation required less machinery and fuel compared, which reduced emissions from fuel combustion and secondary sources. In the study of O'Brien at al. (2012b) a reduction of total GHG emissions per unit of product was achieved reducing on-farm synthetic N fertilizer application (in order to decrease the farm N surplus). The strategy reduced N_2O emissions from on-farm fertilizer use, N_2O from NO_3 leaching and NH_3 re-deposition and off-farm emissions from fertilizer production. Similar results in reducing N input were estimated by Vellinga et al. (2011). On the contrary Belflower et al. (2012) did not find any mitigation effect related to a reduction in the application rate of the inorganic nitrogen fertilizer on grassland of pasture-based dairy, probably because in the model nitrogen availability was reduced, potentially reducing grass yield and protein content. In the same work Belflower et al. (2012) analyzed the effect of removing free stall barns and let all cattle on pasture throughout the year. Convert land currently used for annual ryegrass and corn silage production to perennial pastures use of grazing had a relatively small impact on the carbon footprint because a reduction in milk production. This further illustrates that the use of grazing may not have much impact on the carbon footprint of milk production when compared to confinement feeding.

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WORK 1: ***LIFE CYCLE ASSESSMENT OF MILK PRODUCTION OF 41 INTENSIVE DAIRY FARMS IN NORTH ITALY.***

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ABSTRACT

Agriculture and animal husbandry are important contributors to global emissions of greenhouse and acidifying gases. Moreover, they concur to water pollution and to consumption of non-renewable natural resources, such as land and energy. The Life Cycle Assessment (LCA) methodology allows to evaluate the environmental impact of a process from the production of inputs to the final product and to assess simultaneously several environmental impact categories: among others, GHG emissions, acidification, eutrophication, land use and energy use. The main purpose of this study was to evaluate, with LCA methodology, the environmental impact of milk production in a sample of 41 intensive Italian dairy farms and to identify, among different farming strategies, those associated with the best environmental performances. The functional unit was 1 kg Fat and Protein Corrected Milk (FPCM). Farms showed characteristics of high production intensity: FPCM, expressed as t per hectare, was 30.8±15.1. Total GHG emission per kg of FPCM at farm gate was 1.30±0.19 kg CO_2 eq. The main contributors to climate change potential were: emissions from barns and manure storage (50.1%) and emissions for production and transportation of purchased feeds (21.2%). Average emission of gases causing acidification to produce 1 kg of FPCM was 19.7±3.6 g of SO_2 eq. Eutrophication was 9.01±1.78 PO_4^{3-}eq. per kg FPCM on average. Farms from this study needed on average 5.97±1.32 MJ per kg FPCM from non-renewable energy sources. Energy consumption was mainly due to off-farm activities (58%) associated to purchased factors. Land use was 1.51±0.25 m^2 per kg FPCM. The farming strategy based on high conversion efficiency at animal level was identified as the most advantageous to mitigate environmental impact per kg milk at farm gate, especially in terms of GHG production and non-renewable energy use per kg of FPCM.

Keywords: dairy farm, milk, sustainability, Life Cycle Assessment

1. INTRODUCTION

LCA is a method that allows to assess simultaneously several environmental impacts associated to a product, from energy use to global warming; in its widest meaning, it incorporates into the analysis all processes involved in manufacturing of a product, from raw material extraction to possible waste treatments (de Boer, 2003).

Agriculture and livestock sector are important contributors to global emissions of greenhouse gases (GHG), in particular methane (CH_4) and nitrous oxide (N_2O). Global warming impact of ruminant livestock farms is particularly high due to CH_4 emissions from enteric fermentation and manure storage and handling, and due to the intensive nitrogen (N) cycle leading to direct and

indirect N_2O emissions (Olesen et al., 2006). According to Casey & Holden (2005) enteric fermentation and manure management are responsible for 60% of the global warming potential of milk production.
Agriculture considerably contributes to the release of N compounds to the atmosphere, as NH_3, NOx and N_2O. Especially for ammonia, agriculture and animal husbandry are by far the main sources. N volatilizations occurs during and after production, storage and application of organic (slurry and manure) and mineral fertilizers (Brentrup et al., 2000). N emissions cause acid deposition and intensive acidification of water and soil; a study on German dairy farms showed that acidification impact is almost exclusively caused by ammonia emission from the cattle keeping (Haas et al. 2001).
Eutrophication is an indicator of nutrient enrichment in surface water (van Calker et al. 2004) that is considered the direct cause of the increased plant and microbial growth with consequent consumption of the oxygen dissolved in the water. Contribution of on-farm activities on eutrophication comes mainly from nitrate leaching, phosphate run-off, and ammonia volatilization; in particular ammonia emissions occur during application of fertilizers in the production of on-farm feeds and from manure excreted in the stable, during storage and during grazing. In the conventional dairy farming systems, on-farm feed production contributed 90% and animals contributed 9% to on-farm eutrophication potential. Off-farm eutrophication derived mainly from nitrate leaching, phosphate run-off and ammonia volatilized during application of fertilizers in the production of purchased concentrates and roughages (Thomassen et al., 2008).
Use of natural resources as land, fossil fuels and water represents an important environmental impact category in LCA studies (de Boer, 2003). Land use includes both area needed for fodder production on the farm and area needed for cultivation of purchased fodder (Müller-Lindenlauf et al., 2010). Land use is generally higher in organic dairy farms than in the conventional ones, due to decreased crop yields per ha (Thomassen et al., 2008; Kristensen et al., 2011). The combustion of fossil fuels gives an important contribution to CO_2 emission in the atmosphere from livestock farms. Fossil fuels are primarily used in cultivation of feed, both on-farm and outside the farm, for manure application, transport of animals, processing and transport of feed. Electricity is mainly used on the dairy farm for milking and milk cooling (Flysjö et al., 2011).
In the North of Italy favorable climatic and infrastructural conditions determined, during the last decades, a great concentration of livestock farms with intensive utilization of natural resources (i.e. land, air, water). The Po valley represents altogether only 18% of the utilized agricultural area of the country but accounts for 49% of cattle, 62% of pigs and 63% of poultry population of Italy. The high intensity of land use by animal production farms combined with the high soil vulnerability for nitrogen leaching has created nitrogen pollution problems in the ground and surface water (de Roest, 2000). Many researches were conducted in Italy to find strategies to reduce nitrogen water pollution (Xiccato et al., 2005; Penati et al., 2011) but, at the moment, a very limited number of studies considered the integral environmental performances of the dairy farm in a cradle-to-farm-gate perspective (Penati et al., 2010; Fantin et al., 2012).
The purpose of this study was to evaluate, with LCA methodology, the environmental impact of milk production in a sample of intensive Italian dairy farms located in the Po valley. Another aim was to identify, among different farming strategies, the solution associated to the best environmental performances.

2. MATHERIALS AND METHODS

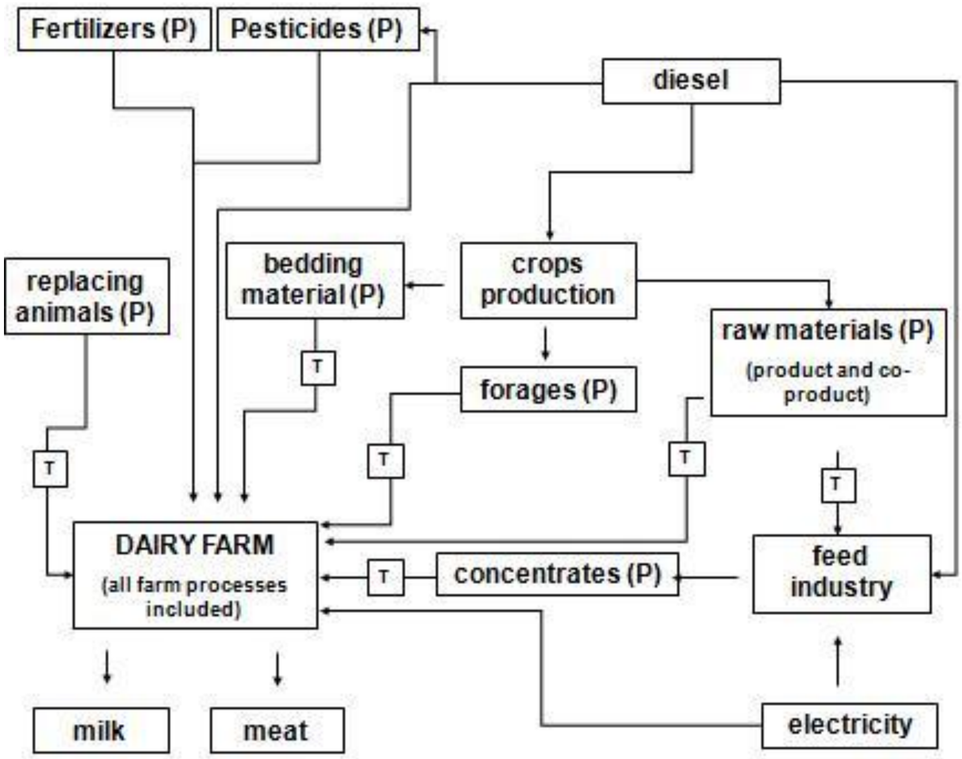
2.1. Goal and scope definition

2.1.1. Objectives

The aim of these study was to evaluate the environmental impact of milk production in a high intensive dairy farms of Lombardy region and to identify some farming strategies in order to mitigate it.

2.1.2. System boundaries

The system under study included the whole life cycle required for the production of raw milk, from the production of inputs to products leaving the farm-gate (cradle to farm gate) , i.e. excluding transport or processing of raw milk. For each dairy farm, a detailed ‘‘cradle-to-farm-gate’’ LCA was performed. In particular we considered all the processes related to the farms activity i.e. forages and corps production, energy use, fuel and electricity consumption, manure and livestock management. The external factors (input) like production of fertilizer, production of pesticides, production of fodders and raw materials, production of concentrate feed, production of electricity and fuel, breeding of replacing animals and production of litter materials (straw and sawdust) were considered parts of the system. Related transport associated with the production of purchased inputs (only livestock’s feed) was included (Thomassen et al., 2008). All data were referred to year 2009.



T: transportation included
P: purchased

Fig. 1 - System boundaries.

2.1.3. *Functional unit*

The functional unit chosen was 1 kg fat protein corrected milk (FPCM) leaving the farm gate (Thomassen et al., 2008). FPCM is a correction factor that considers both the fat and the protein content of the milk. All milk was converted to FPCM with 4.0 % fat and 3.3 % protein, using the formula: FPCM (kg) = (0.337 + 0.116 x % fat + 0.060 x % protein), from Gerber et al. (2010).

2.1.4. *Allocation*

In these study an economic allocation was used for each farm considering the value of the two main product sold during the year 2009: milk and meat. Other product sold like fodder or grains and in some cases manure were not taken into account for their small value. The mean values for the economic allocation in our study was 94.2% for milk and 5.8% for meat (± 2.71%), these data show the high specialization in milk production for these farms. Economic allocation (Table 1) was also used to distribute the environmental burden from the production of purchased concentrate feed (Cederberg and Stadig, 2003).

Tab. 1 - Allocation for feed products.

CO-PRODUCT	ECONOMIC ALLOCATION %	Main-product	Reference
SOYBEAN MEAL	59.3	Soybean meal	Jungbluth et al., 2007
SOYBEAN OIL	40.7	Soybean meal	Jungbluth et al., 2007
SOYBEAN HULLS	2.3	Soybean meal	Associazione Granaria di Milano, 2012
BEET PULP	15.0	Sugar	Cederberg and Flysjö, 2004
BEET MELASSOES	5.0	Sugar	Cederberg and Flysjö, 2004
PALM OIL	97.3	Palm oil	Cederberg and Flysjö, 2004
MAIZE GLUTEN MEAL	2.9	Bioethanol	Buratti et al., 2008
MAIZE GERM MEAL	9.8	Bioethanol	Buratti et al., 2008
MAIZE DDGS	18.0	Bioethanol	Börjesson, 2008
WHEAT DDGS	18.0	Bioethanol	Börjesson, 2008
SUNFLOWER MEAL	37.0	oil	anonimous
WHEAT STRAW	24.0	Grains	Associazione Granaria di Milano
WHEAT MILL RUN	1.1	Flour	CCIAA Lodi, 2012
WHEAT MIDDS	5.3	Flour	CCIAA Lodi, 2012
WHEAT BRAN	5.3	Flour	CCIAA Lodi, 2012
RAPE MEAL	25.7	Rape oil	Jungbluth et al., 2007
COTTON SEED	12.0	Cotton	www.pre.nl/download/HandoutsDay1.pdf
SAW MEAL*	mass allocation	Wood	Milota et al., 2004

* for bedding

2.1.5. *Impact categories*

The environmental impact categories chosen for the study were:

- Global warming potential (kg CO_2 eq. 100-year horizon)
- Acidification (g SO_2 eq.)
- Eutrophication (g PO_4^{3-} eq.)
- Non-renewable energy use (MJ eq.)
- Land use (m^2)

2.2. Inventory analysis

2.2.1. Data sources

The presented study was based on a survey of 41 dairy farms in Lombardy region, all farms are members of a cooperative feed factory. This group of farm might be considered representative of the dairy farming system of the Po Valley.

To the selected farmers were requested to answer a questionnaire in order to collect all the farm data that are necessary to carry on this type of analysis. Data were collected on a number of dairy cows and young stock, animal production (milk and meat delivered), milk quality, area of grassland and other crops (fodder and grains), amount of fertilizer and pesticide used, roughages and concentrate feed purchased, bedding material purchased, feed strategies (ration formulation), manure management and energy consumption (electricity, diesel and gas). As in previous studies, the environmental burdens associated with the production and consumption of substances used for disinfection and cleaning maintenance of the stables and milking systems were not considered due to the lack of data (Cederberg and Mattson, 2000; Casey and Holden, 2005; Thomassen et al., 2008). However, these environmental burdens are expected to be insignificant (Hospido et al., 2003). Data related to concentrate feed purchased and ration formulation (especially for lactating animals) were supplemented with other more information taken directly from the feed factory archives. The Table 2 reports a summary of the information collected during the farm visit.

Tab. 2 - Farm information.

LAND and CROP MANAGEMENT	LIVESTOCK and MANURE MANAGEMENT	ANIMAL FEEDING	ENERGY CONSUMPTION	FARMS PRODUCTS
Area (ha)	Number of cows	Feeding systems	Electricity use (KWh $year^{-1}$)	Milk delivered (kg $year^{-1}$)
Type of crop/forage	Number of young animals	Feed ingestion (kg feed d^{-1})	Diesel use (kg $year^{-1}$)	Meat delivered (number and type of animals sold in the year)
Yeld (kg ha^{-1})	Housing system	Rations composition	Methane and LPG use (m^3 $year^{-1}$)	Milk quality (% Protein and % Fat)
Pesticides (kg ha^{-1})	Manure storage system and management	Purchased concentrate feeds (ton $year^{-1}$) and their origin		
Chemical fertilizers (kg ha^{-1})	Manure exported (ton $year^{-1}$)	Purchased forages (ton $year^{-1}$) and their origin		
Organic fertilizers (kg ha^{-1})	Purchased bedding materials (ton $year^{-1}$) and their origin			
Land operations	Number of purchased replacing animals and their origin			

Data concerning concentrate feed production were requested to the feed factory, so it was possible to know exactly the composition of the different concentrates (more than 35), the origin of raw materials and the sea or road trip from the place of origin to the feed factory and from the feed factory to the farms. In these work the data referred to the production of crops (and their by-products) used in feed formulation were collected from different sources. When it was possible the processes from Ecoinvent (2007) database (Prè Consultants - Simapro PhD 7.3.3, 2012) were considered and slightly adapted to the different conditions (mainly related to the country of origin and the transport distances) but when no data were available from the Ecoinvent (2007) database other sources were taken into account in the way to supply as much as possible to the lack of data. Inventory of raw materials used in production of feeds is resumed in Table 3.

Tab. 3 - Raw materials used in commercial feed production, their origin and rferences for inventory data.

RAW MATERIAL	ORIGIN	REFERNCE
barley meal	Germany 75%	Nemecek and Kägi, 2007
barley meal	Italy 25%	Nemecek and Kägi, 2007
beet pulp	France 30%	Jungbluth et al., 2007
beet pulp	Italy 70%	Jungbluth et al., 2007
corn DDGS	North Est Europe 50%	Jungbluth et al., 2007
corn DDGS	Italy 50%	Jungbluth et al., 2007
corn germ meal	Italy	Jungbluth et al., 2007
corn gluten meal	Italy	Jungbluth et al., 2007
corn meal	Hungary 55%	Nemecek and Kägi, 2007
corn meal	Italy 45%	Baldoni e Giardini, 2002; Ribaudo, 2002; data from this work
cottonseed	Greece	Nemecek and Kägi, 2007
milk powder	France	Nielsen, 2003a.
molasses	Thailandia	Jungbluth et al., 2007
palm oil	Malaysia-Indonesia	Hischier, 2007
rapeseed meal	Germany	Jungbluth et al., 2007
soybean	Argentina	Jungbluth et al., 2007
soybean hulls	USA	Jungbluth et al., 2007
soybean meal	Italy 80%	Jungbluth et al., 2007
soybean meal	Argentina 20%	Jungbluth et al., 2007
soybean oil	Italy	Jungbluth et al., 2007
sunflower meal	Ucraine	Ragaglini et al., 2011
wheat bran	Italy	Nielsen and Nielsen, 2003
wheat DDGS	Italy 50%	Jungbluth et al., 2007
wheat DDGS	Austria 50%	Jungbluth et al., 2007
wheat meal	Italy 40%	Nemecek and Kägi, 2007
wheat meal	Austria 60%	Nemecek and Kägi, 2007
wheat other co-products	Italy	Nielsen and Nielsen, 2003

2.2.2. On-farm emissions estimation

The total emissions (into the atmosphere and in the soil) at farm level were estimated. Methane emissions from enteric fermentation of livestock and emissions from manure management (manure handling and storage) in form of methane, nitrous oxide and ammonia were taken into account. Moreover during the application of organic and mineral fertilizers on cultivated soils occur emissions in the air in form of ammonia (NH_3) and nitrous oxide (N_2O) besides water emissions consist of nitrates (NO_3) and phosphate (PO_4^{3-}). Emissions generated by combustion of fuels were considered. Emissions from livestock respiration are part of a rapidly cycling biological system, where the plant matter consumed was itself created through the conversion of atmospheric CO_2 in organic compounds. Since the emitted and the absorbed quantities are considered to be equivalent, livestock respiration is not considered to be a net source under the Kyoto Protocol (Steinfeld et al., 2006). The variation in soil carbon stock in the farm land wasn't considered in this work, although emissions due to land use change can be substantial, their quantification and allocation to commodities is conceptually and methodologically difficult (Lesschen et al., 2012).

2.2.2.1. Greenhouse gases (GHG) emissions

Methane (CH_4) emissions from livestock enteric fermentations were estimated on the base of dry matter intake (DMI) using the equation: CH_4 (MJ d^{-1}) = 3.23 (± 1.12) + 0.809 (± 0.0862) × DMI (kg d^{-1}) from Ellis et al. (2007) (equation 2d for dairy cows, R^2=0.65) and considering the factor 55.65 (MJ kg^{-1} CH_4) the energy content of methane (IPCC 2006a).

Methane emissions from manure management were estimated using Tier 2 method suggested from IPCC (2006a). This estimation is based on the volatile solid excretion rate (VS), to obtain this value is necessary to know which is the gross energy (GE) of the animal diets. The GE of the diets (kj kg^{-1} DM) was calculated using the equation from Shiemann (1988) which considers the diets components (crude protein (CP g kg^{-1} DM), ether extract (EE g kg^{-1} DM), crude fiber (CF g kg^{-1} DM) and nitrogen free extract of the diet (NfE g kg^{-1} DM). Different manure management methane emission factors (MCFs) were considered for the different animal categories on the base of housing systems and manure handling. The model used to estimate methane emissions form storages is represented in Table 4.

Tab. 4 - Equations and emission factors for the estimation of methane emissions at storages farms level.

Amount	Emission Factor	Reference
CH_4 = VS * B_0 * 0.67 * MCF/100 * MS		Eq. 10.23 - IPCC (2006a)
VS = [GE*(1-DE/100)+(UE*GE)]*[(1-Ash)/18.45]		Eq. 10.24 - IPCC (2006a)
GE(kj/kg) = 23.9 * CP + 39.8 * EE + 20.1 * CF + 17.5 * NfE		Shiemann (1988)
DE: feed digestibility	75% for dairy cows (Table 10A.1 North America value) and 65% for growing heifers/steers (Table 10A.2 North America value)	IPCC (2006a)
UE: urinary energy	0.04 *GE	IPCC (2006a)
Ash: the ash content of manure calculated as a fraction of the dry matter feed intake	0.08	IPCC (2006a)
B_0: Maximum methane-producing capacity of the manure	0.24$m^3 CH_4$/kg VS for dairy cows (Table 10A-4: Western Europe) 0.18$m^3 CH_4$/kg VS for heifers (Table 10A-5: North America) 0.1$m^3 CH_4$/kg VS for calves (Table 10A-5: Middle East)	IPCC (2006a)
MCF: manure management methane emission factors (annual average temperature = 15°C)	MCF solid storage: 4 MCF liquid slurry: 17 MCF pit storage: 27	IPCC (2006a)
MS: fraction of livestock category manure handled using manure management system S		IPCC (2006a)

Animal N excretion was estimated as proposed by the IPCC (2006a) Tier 1 method considering default values for nitrogen excretion rate (Nrate) and for typical animal mass for livestock category (TAM).

Emissions of nitrous oxide (N_2O) may occur directly and indirectly during manure management (storages and spreading), to estimate direct nitrous oxide emissions from manure management systems were used the Tier 1 method from IPCC (2006a), also in N_2O emissions estimation were considered different default emission factors (EF) in the same way of methane emissions. Indirect emissions of N_2O from manure storages, which result from volatile nitrogen losses, occur primarily in the forms of NH_3 and NOx and were considered to be insignificant (Castanheira et al., 2010). In Table 5 are reported the models used to estimate nitrous oxides losses that occurred during manure storages.

Tab. 5 - Equations and emission factors for the estimation of nitrous oxide emissions at storages level.

Pollutant	Amount	Emission Factor	Reference
N_2O direct	**N_2O = Nex * MS * EF * 44/28[a]**		Eq. 10.25 - IPCC (2006a)
	Nex= Nrate * TAM /1000 *365		Eq. 10.30 - IPCC (2006a)
	Nrate: default N excretion rate	dairy cattle: 0.44 kg N (1000 kg animal mass)$^{-1}$ day^{-1} (Table 10.19: North America) other cattle: 0.31 kg N (1000 kg animal mass)$^{-1}$ day^{-1} (Table 10.19: North America)	
	TAM: typical animal mass for livestock category	604 kg for dairy cows (Table 10A-4: Western Europe) 389 kg for heifers (Table 10A-5: North America) 173 kg for calves (Table 10A-5: Middle East)	
	MS: fraction of livestock category manure handled using manure management system S		
	Emission factors (Table 10.21)	EF solid storage: 0.005 (0.0027 - 0.01) EF liquid slurry: 0.005 EF pit storage: 0.002	

[a]44/28: conversion factor (molecular weights) from N-N_2O to N_2O.

Direct and indirect N_2O losses from fertilizers application were estimated following the Tier 1 method suggested from IPCC (2006b) but considering only the amount of nitrogen applied to the soils from synthetic fertilizers an manure (slurry and solid) as explained in Table 6.

Tab. 6 - Equations and emission factors for the estimation of nitrous oxide emissions at field level.

Pollutant	Amount	Emission Factor	Reference
N_2O direct	**N_2O = (Nsn + Non) * EF * 44/28[a]**		Eq. 11.2 IPCC (2006b)
	Nsn: annual amount of synthetic fertilizer N applied to soils		
	Non: annual amount of managed animal manure applied to soil		
	Emission factors (EF_1 - Table 11.1)	EF_1: 0.01 (0.003 - 0.03)	
kg N_2O indirect	**$N_2O_{(ATDN)}$ = [(Nsn * Frac_GasF) + (Non * Frac_GasM)]*EF * 44/28[a]**		Eq. 11.9 IPCC (2006b)
	Frac_GasF: fraction of synthetic fertilizer N that volatilizes as NH_3 and NOx, kg N volatilized (kg of N applied)$^{-1}$ (Table 11.3)	Frac_GasF: 0.1 (0.03 - 0.3)	
	Frac_GasM: fraction of applied organic N fertilizer materials that volatilizes as NH_3 and NOx, kg N volatilized (kg of N applied)$^{-1}$ (Table 11.3)	Frac_GasM: 0.2 (0.05 - 0.5)	
	Emission factors (EF_4 - Table 11.3)	EF_4: 0.01 (0.002 - 0.05)	
	$N_2O_{(L)}$ = (Nsn + Non) * Frac_Leach * EF * 44/28[a]		Eq. 11.10 IPCC (2006b)
	Frac_Leach: fraction of all N added in managed soils that is lost through leaching and runoff, kg N (kg of N additions)$^{-1}$ (Table 11.3)	Frac_Leach: 0.3 (0.1 - 0.8)	
	Emission factors (EF_5 - Table 11.3)	EF_5: 0.0075 (0.0005 - 0.025)	

[a]44/28: conversion factor (molecular weights) from N-N_2O to N_2O.

CO_2 emissions from fuel combustion were estimated on the base of fuel consumption for each farm excluding the quota of fuel used in crops operation (i.e. plowing, harrowing, fertilizing, chopping, etc.), the emission factor used are those proposed by Cederberg (1998): 3.04 kg CO_2 l^{-1} diesel used.

2.2.2.2. Other emission on farm level

According to Castanheira et al. (2010) and Zehetmeier et al. (2011) nitrogen volatilization that occur from animal husbandry and manure storages was considered to be in the form of ammonia (NH_3) . In this case, NH_3 emissions were obtained by multiplying the amount of nitrogen excreted by the fraction of volatile nitrogen (Frac_GasMS) as indicated in Tier 1 method of IPCC (2006a) and summarized in Table 7.

Tab. 7 - Equation and emission factors for the estimation of ammonia emissions at storages level.

Pollutant	Amount	Emission Factor	Reference
NH_3	**Nvolatilization: Nex * MS * Frac_GasMS/100 *17/14[a]**		Eq. 10.26 - IPCC (2006a)
	Frac_GasMS: N loss from MMS due to volatilization of N-NH_3 and N-NOx (Table 10.22 – dairy cows)	Frac_GasMS solid storage: 30% (10 – 40)	
		Frac_GasMS liquid slurry: 40% (15 – 45)	
		Frac_GasMS pit storage: 28% (10 – 40)	
	MS: fraction of livestock category manure handled using manure management system S		

[a]17/14: conversion factor (molecular weights) from N-NH_3 to NH_3.

NH_3 and NOx emissions on field level were calculated respectively as 0.084 (0.06 – 0.1) and 0.026 (0.05 – 0.104) kg kg^{-1} fertilizer-N applied (both manure and synthetic fertilizers) on the base of the EEA (2009) Tier 1 method.Nitrogen leaching occurs in form of NO_3. For foreign crops and national crops (excluding luzerne, grassland and sugarbeet) it was calculated multiplying the amount of nitrogen applied on field by the default value (0.3 (0.1 - 0.8) kg N kg N^{-1} additions) of nitrogen lost by leaching and runoff proposed by IPCC (2006b) and then converted in NO_3 using the molecular weights ratio (NO_3/N = 62/14). To evaluate nitrogen leaching during production of national luzerne, grassland and sugarbeet were used respectively the emission factors 0.087; 0.015 and 0.13 (Penati, 2009) that express kg of N leached per kg of N input. To estimate emissions of PO_4^{3-} were considered the amount of phosphorus loss in dissolved form to surface water (run-off) and leaching as proposed by Nemecek and Kägi, (2007), and converted in phosphate (the coefficient of 95/31 is used in order to express the results in kg of PO_4^{3-} instead of P. This method considers the amount of phosphorus excreted by the animals and applied to the field and also the input from chemical fertilizers. The proportion of phosphorus present in solid manure or liquid slurry was calculated on the bases of LU confinement system. These method takes also into account the quota of land that is arable or meadow. Phosphorus loss in particulate form soil erosion to surface water wasn't take into account because lack of data. The models applied is detailed in Table 8.

Tab. 8 - Equation and emission factors for the estimation of phosphorus emissions at field level.

Pollutant	Amount	Emission Factor	Reference
PO_4^{3-}	**P_{gw} (quantity of P leached to ground water) = P_{gwl} * F_{gw}**		Nemecek et al. (2007)
	P_{gwl} = average quantity of P leached to ground water for a land use category (kg P ha $year^{-1}$)	P_{gwl} arable land: 0.07 P_{gwl} permanent pasture and meadow: 0.06	
	F_{gw}: correction factor for fertilization by slurry	1+0.2/80 * P_2O_{5sl}	
	P_2O_{5sl} is the quantity of P_2O_5 contained in the slurry		
	P_{ro} (quantity of P lost through run-off to rivers) = P_{rol} * F_{ro}		
	P_{rol} = average quantity of P lost through run-off for a land use category (kg P ha $year^{-1}$)	P_{rol} open arable land: 0.175 P_{rol} intensive permanent pasture and meadow: 0.25 P_{rol} extensive permanent pasture and meadow: 0.15	
	F_{ro}: correction factor for fertilization with P = 1 + F_{ro_min} + F_{ro_sl} + F_{ro_man}		
	F_{ro_min}: correction factor for fertilization by mineral fertilizers P_2O_{5min} is the quantity of P_2O_5 contained in the mineral fertilizer	0.2/80 * P_2O_{5min}	
	F_{ro_sl}: correction factor for fertilization by slurry	0.7/80 * P_2O_{5sl}	
	F_{ro_man}: correction factor for fertilization by manure	0.4/80 * P_2O_{5man}	
	$P_2O_{5ro\_man}$ is the quantity of P_2O_5 contained in the manure		

2.2.3. *Off-farm emissions estimation*

The off-farm emissions are mainly related to the production chain of commercial feed, from the cultivation of crops to the arrival of the final commercial product to the farm, including processing of raw materials and transport. In the estimation of off-farm emissions was also considered the production of roughages and the bedding material purchased and the transportation related, the

production of chemical fertilizers and pesticides but not the related transportation, the production of diesel and electricity and the process to rear the replacing animals purchased by the farms.
The estimation of the emissions that occur during the production chain of commercial feed was carried out with the assistance of the Simapro PhD 7.3.3 (Prè Consultants, 2012) software. The transport distances were calculated on the base of information given from the feed factory using http://www.viamichelin.it for road trips and http://sea-distances.com/ for the ship transport. The energy consumption for processing 1 ton of livestock feed (about 12% water content) at the factory's gate is considered equal to 53.5 kWh and 140 MJ from heat according to Nielsen (2003b).
Production of forages purchased by the farms from the local market were taken into account. As described before, emissions on the field level were estimated on the base of data collected in the survey integrated with the data concerning crops operation (i.e. plowing, harrowing, sowing, harvesting, etc.) taken from Econivent (2007) database.
In Table 9 are reported the crops and the source considered for forages production. Also the distance covered by the products from the place of origin to every single farm was taken into account.

Tab. 9 - Raw materials used in commercial feed production and their origin.

FORAGES PURCAHSED	SOURCES
grass hay	data from this work;
ryegrass silage	Baldoni and Giardini, 2002; data from this work
corn silage	Baldoni and Giardini, 2002; Ribaudo, 2002; data from this work;
luzerne hay	Baldoni and Giardini, 2002; data from this work
wheat straw*	Nemecek and Kägi, 2007
Crops operation	Nemecek and Kägi, 2007

* also for bedding material

Fertilizers used by the farms were mainly urea, ammonium nitrate, di-ammonium phosphate and potassium carbonate. From the total amount declared by the farmers were calculated the real quantity of the nutrients applied to the soil on the base of the share included in the commercial product. Emission related to the production of fertilizers (per kg of nutrients) were estimated using the information present in Simapro PhD 7.3.3 (Prè Consultants, 2012) databases in particular from Patyk and Reinhardt (1997) and Nemecek and Kägi (2007).
More than 50 active substances (both herbicides and insecticides) were used in crops operation by group of farms studied. Knowing the name of the commercial products and the quantity spread on field an estimation of the active substances really used was done on the base of what declared on the products label. Emissions evaluation was performed using information from Nemecek and Kägi (2007).
Data used for estimation of energy production were taken from Jungbluth (2007), Frischknecht et al. (2007) and Nemecek and Kägi (2007). Data related to transportation were taken from Spielmann et al. (2007).
7 farms used to buy some replacing animals from other national breeding, generally located in the same territory of the farms under study. Emissions from the rearing process of the purchased animal should be considered, for that reason, on the base of information collected during the interview we assumed:

- age of the purchased animal: 24 months
- feed ingestion: an average value calculated on the base of data collected during this work

- housing system: on litter (from 0 to 12 months) in order to produce a solid manure and on concrete floor (from 12 to 24 months) in order to produce a liquid slurry.

4 farms purchased some replacing animals from foreign countries, mainly from North Germany, for that reason we considered all the animals buy out of Italy coming from North Germany. Less information were available about German farms, some data were recovered from http://www.rinderpraxis.com/, Delagarde et al. (2011), Zehetemeir et al. (2011) and some anonymous sources. Some basic assumption were done in the way to supply the lack of data:

- age of the purchased animal: 24 months
- feed ingestion: based on an anonymous document
- housing system: on litter (from 0 to 6 months) in order to produce a solid manure and on slatted floor (from 6 to 24 months) in order to produce a liquid slurry.

In both cases (national and foreign heifers) enteric methane emissions were estimated and also emissions from manure and slurry storages. In the model are considered also the emissions related to the production of feed (concentrates and roughages) consumed by these animals. In the end emissions produced during the transport from the place of origin to the farms were taken into account.

2.3. Impact assessment

Life Cycle Assessment was carried out with the assistance of a commercial LCA software package, Simapro PhD 7.3.3 (Prè Consultants, 2012). It is an open structure program that can be used for different types of life cycle assessments (Cotana et al., 2010). In particular in the evaluation of GWP, Eutrophication, Acidification an Non-renewable energy use was used the EPD 1.03 (2008) method. Land occupation was evaluated using Ecological footprint (2009) method. Both the two methods are present in the software database.

2.4. Farm nutrient balance

According to Schröder et al. (2003), the nutrient surplus as determined by a farm-gate balance (FGB) of inputs and outputs can be considered a good tool to evaluate the nutrient flows at the farm scale and improving nutrients management.

In this work only N and P balance were performed (Steinshamn et al., 2004; Nielsen and Kristensen, 2005; Thomassen and De Boer, 2005; Virtanen and Nousiainen, 2005; D'Haene et al., 2006; Giustini et al., 2007; Fangueiro et al., 2008; Segato et al., 2009; Penati et al., 2011) but it may be valuable to include other elements of interest for animal production and dairy, such as potassium (Öborn et al., 2003). The inputs and outputs used for calculating the farm nutrient balance in each farm were defined according to the study by Schröder et al. (2003) and are described in Table 10. Also this data were collected during the farm visit, the raw information about input and output were then adjusted using the conversion factors (for N and P). N inputs from irrigation water were not considered but the amount of atmospheric N fixed from crops (especially legumes) were taken in account. The farm-gate nutrient surplus was calculated as the difference between input and output of nutrient divided by the agricultural area at the farm scale. Schröder et al. (2003) considered that N gaseous losses have also to be taken into account to achieve a more detailed diagnostic of the nutrient losses at farm scale. Therefore, the N gas emissions, mainly NH_3 volatilization, from animal housing, storage and spreading of slurry have to be estimated (Fangueiro et al., 2008). In this work gaseous losses were considered as 28% of total nitrogen surplus, this rate is reported in DM 7/4/2006 (MIPAF, 2006).

Tab. 10 - Inputs and outputs used for calculating farm-gate nutrient balance in each farm.

INPUT	ESTIMATION
Concentrate feed (commercial feed, grains and by-products)	N content: (% PG on fresh matter/100 * kg concentrate feed)/6.25
	P content: (% P on fresh matter/100 * kg concentrate feed)
Purchased forages	N content: (% PG on fresh matter /100 * kg concentrate feed)/6.25
	P content: (% P on fresh matter/100 * kg concentrate feed)
Mineral fertilizer	N content: % N/100 * kg fertilizer
	P content: (% P_2O_5/100 * kg fertilizer) * 0.4364
Purchased animals	N content: kg LW * 2.9/100 [a]
	P content: kg LW * 0.7/100 [a]
Purchased bedding material (straw or sawdust)	N content: (kg straw * 0.46/100 + kg sawdust * 0.06/100)
	P content: (kg straw * 0.092/100 + kg sawdust * 0.008/100)
Atmospheric deposition [b]	N deposited: ha farm land * 20 kg N [c]
Crops fixation [b]	N fixed meadow: ha meadow * 15 kg N [d]
	N fixed lucerne: ha lucerne * 200 kg N [e]
OUTPUT	**ESTIMATION**
Milk	N content: (% PG milk/100 * kg milk sold)/6.38
	P content: (% P milk/100 * kg sold)
Animals (live and dead animals)	N content: kg LW * 2.9/100 [a]
	P content: kg LW * 0.7/100 [a]
Sold forages and grains	N content: (% PG on fresh matter/100 * kg product sold)/6.25
	P content: (% P on fresh matter/100 * kg product sold)
Manure (solid and liquid)	N content: (kg manure * 0.34/100 + kg slurry * 0.38/100)
	P content: (kg manure * 0.07/100 + kg sawdust * 0.105/100)

[a] N content of LW is 2.9% and P content of LW is 0.7% (Cornell, 2008).

[b] Inputs considered only for N.

[c] D.M. 1999 (MIPAF, 2006)

[d] Penati et al., 2011.

[e] Baldoni and Giardini, 2002.

2.5. Evaluation of economic indicators

During the farm visit was asked to the farmers to answer also to some questions about the economic trend of the farm in order to collect some information useful to analyze some economic indicators. Our intention was first of all estimate the environmental impact of each farm and then perform an economic evaluation of the farm activity.

Economic data collected are shown in Table 11 and are related to all the voices of variable costs and revenues. Fixed costs (like rent of the land, depreciation of machineries and building, salary of the workers, etc…) were not considered.

Tab. 11 - Variable costs and revenues asked to each farmer. The reference year is 2009.

COST	REVENUES
Concentrate feed (commercial feed, grains and by-products)	Milk [a]
Purchased forages	Animal sold (for slaughering and for "life")
Pourchased bedding material (straw or sawdust)	Forages and grains sold
Mineral fertilizer	PAC contributes
Pesticides and isecticides	Other contributes
Pourchased animals	Manure sold
Veterinary service and medicines, semens, cleaning products, etc…	
Disposal of dead animals	
Energy (fuels and electricity)	
Water (for the barn and for irrigation)	

[a] the prices for milk quality are included

The economic indicators estimated in this work were:

- Gross margin of a forage based livestock enterprise: is Output from the enterprise less the Variable Costs, including the allocated variable costs of grass and other forage (DEFRA, 2010).
- Income over feed costs (IOFC) is a popular value as it provides a benchmark for a herd or groups of cows reflecting profitability, current feed prices, and actual milk prices (Hutjens, 2007). The IOFC per cow per day was estimated like: [(kg milk cow^{-1} day^{-1}) * milk price (€ kg^{-1} milk)] - feed cost cow^{-1} day^{-1}.

This evaluation could be done also at the herd level. The IOFC cost implies the estimation of other indicators like feed cost per cow per day that does not reflect milk yield, stage of lactation, or nutrient requirements and feed cost per unit of dry matter that is a useful term when comparing similar regions, breeds, and levels of milk production (Hutjens, 2007). IOFC is certainly strongly influenced by the price of milk and feedstuff but also by the capacity of the animal to convert feed ingested in milk. For that reason also the animal feed efficiency was evaluated. Feed efficiency can be defined as unit of milk produced per unit of dry matter intake (DMI) consumed (Hutjens, 2007).

2.6. Statistical analysis

The SimaPro output sheet of each farm was exported in Microsoft Excel than a statistical analysis were performed using SAS V8 software (SAS, 2001). Statistical analysis was performed using SAS 9.1 software (SAS, 2001). According to Kristensen et al. (2011) the most representative variable was chosen to identify and describe different farming strategies. Five different factors were identified using PROC FACTOR analysis (SAS, 2001). All variables that were represented in the factor pattern with a loading value higher than 0.58 were used to define the farming strategy corresponding to each of the five factors identified. A GLM (SAS, 2001) analysis was performed to evaluate the effect of farming strategies on impact categories of LCA. The percentage of variation for impact categories explained by each factor was calculated as the ration between mean square and the sum of squares of the model. The 41 dairy farms in the sample were classified using a distribution analysis (quantiles) (SAS, 2001) on the basis of the value of each impact category and a GLM analysis was performed taking into account the upper quartile (the 75th percentile) and the lower quartile (the 25th percentile).

Database was also analyzed using the CLUSTER procedure (SAS, 2001). In order to identify different farming systems the following variables were considered: gross margin, feed self-sufficiency, dairy efficiency and stocking density. A GLM analysis was performed on the groups obtained from the CLUSTER procedure.

3. RESULTS AND DISCUSSION

3.1. Dairy farms description

The farms had characteristics of high intensity both in terms of cropping systems and animal production. They had on average 140±81 adult cows and 45.4±28.8 ha of farm land. Average stocking density, expressed as Livestock Units (LU) per hectare (DEFRA, 2010), was high (5.45±2.69) and milk production per cow was on average 10.3±1.2 t FPCM per year. As a consequence, production intensity, expressed as t FPCM per hectare, was very high (30.8±15.1) compared to the results from other European studies: 13.1 t ha^{-1} and 14.4 t ha^{-1} in two recent Italian studies (Penati et al., 2010; Fantin et al., 2012); 10.6 t ha^{-1} in a Dutch analysis (Thomassen et al., 2008). Maize was grown both for silage and for grain in all the farms and occupied, on average, 25.1% and 6.9% of farm land, respectively. On average, 51.3±5.21% of the total dry matter (DM) of cow rations consisted of forages, with 32.9±6.68% of maize silage. Feed self-sufficiency of the farms, was generally low: 58.6±15.7% on total DM consumed. Dairy efficiency was 1.3±0.14 kg FPCM per kg of DM intake, whereas efficiency of nitrogen utilization for milk production as percentage of nitrogen ingested was 27.3%.

3.2. Environmental impact

3.2.1. Climate change

Total GHG emission for milk production at the farm gate was 1.30±0.19 kg CO_2 eq. per kg FPCM (Table 12).

Table 12 - Annual emission of greenhouse gasses (kg CO_2 eq.), Acidification (g SO_2 eq.), Eutrophication (g PO_4^{3-}eq), Energy use (MJ) and Land use (m^2) in 41 dairy farms.

	Mean	SD	Minimum	Maximum
Climate change (kg CO_2 eq. per kg FPCM)				
Total	1.30	0.19	1.02	1.96
On farm	0.91	0.15	0.67	1.47
Off farm	0.38	0.09	0.15	0.64
Acidification (g SO_2 eq. per kg FPCM)				
Total	19.7	3.6	13.7	31.9
On farm	16.6	3.6	10.4	29.7
Off farm	3.11	1.87	1.19	10.9
Eutrophication (g PO_4^{3-}eq. per kg FPCM)				
Total	9.01	1.78	5.99	13.8
On farm	7.06	2.02	3.89	12.4
Off farm	1.93	0.98	0.76	6.11
Energy use (MJ per kg FPCM)				
Total	5.97	1.32	3.69	10.9
On farm	2.15	0.74	1.27	5.07
Off farm	3.44	0.79	1.35	5.58
Land use (m^2per kg FPCM)				
Total	1.51	0.25	0.95	2.06
On farm	0.71	0.26	0.29	1.47
Off farm	0.80	0.24	0.32	1.71

Most of greenhouse gases were produced on-farm (77%). The average global warming potential per kg of milk was the same reported by Haas et al. (2001) for intensive dairy farms in Germany and by Fantin et al. (2011) for the farm phase of dairy chain in Italy. Moreover it was very similar to the results obtained by authors from other European countries (Casey & Holden, 2005; Kristensen et al., 2011; Thomassen et al., 2008). The main components of climate change impact (Figure 2) were: emissions from barns, manure storage and handling (50.1%) and emissions for production and transportation of purchased concentrate feeds (21.2%). The contribution to the climate change potential of the three greenhouse gases considered was on average 59.0% for CH_4, 35.6% for CO_2 and 19.7% for N_2O.

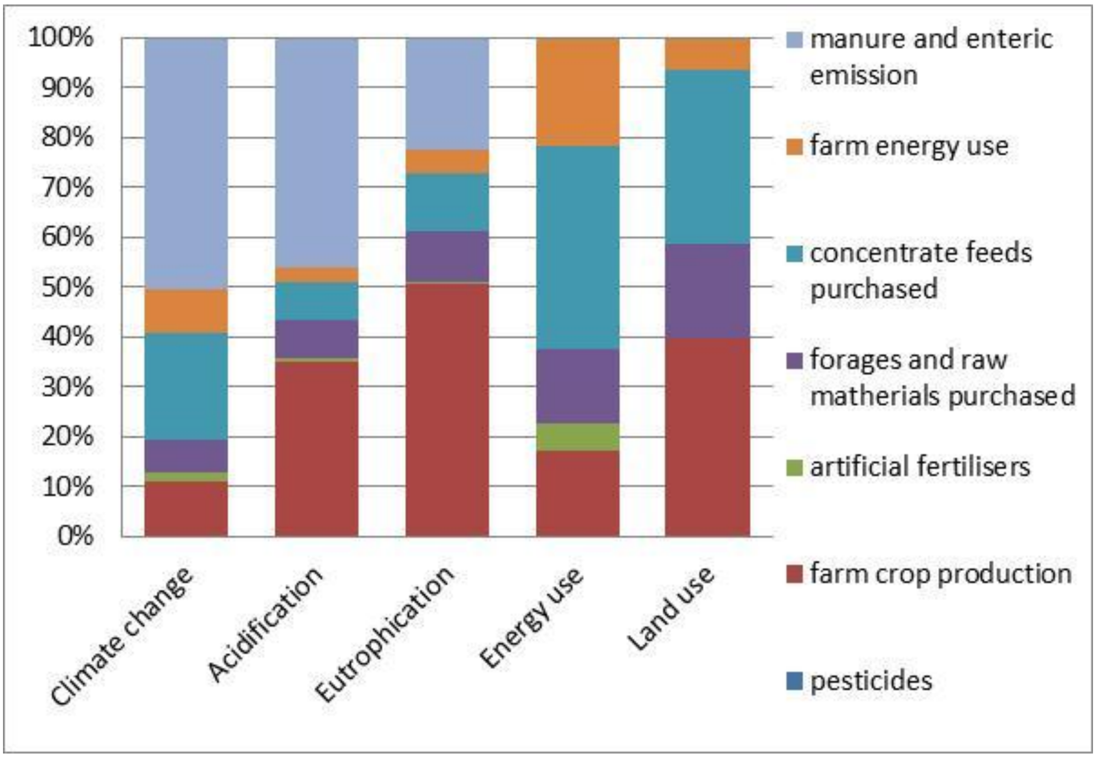


Fig. 2 - Contribution of different compartments (on- and off-farm) to environmental impact.

3.2.2. Acidification

Average emission of gases causing acidification (expressed as SO_2 equivalents) to produce 1 kg of FPCM was 19.7±3.6 g. This result was mainly due to on-farm gas production (84.3%). In particular the main contributors were emissions from barns and manure storage and handling (45.9%) and emissions for feed production on-farm (35.2%). Acidification potential was mainly due to ammonia emission in the air (90.4±1.74%). The average emission of acidifying gases was similar to the results reported by Castanheira et al. (2010) and by Cederberg and Mattsson (2000). Fantin et al. (2012) estimated a contribution of the farm phase in Italian dairy chain to acidification potential of nearly 20 g SO_2 eq. per kg of milk. In contrast, Thomassen et al. (2008) reported much lower acidification potential both in conventional and in organic dairy farms.

3.2.3. Eutrophication

The production of 1 kg of FPCM caused the loss of 9.01±1.78 g on average of phosphate equivalents. Fantin et al. (2012) reported that the farm operation phase contributes to eutrophication with 7.8 g PO_4^{3-} eq. per kg of milk. Eutrophication resulted mainly from on-farm activities (78%). In particular it was mainly due to the application to the soils of artificial fertilizers and manure for feed production at the farm (50.7%), followed by ammonia emission from barns and during manure storage and handling (22.4%). According to this, the main contribution to eutrophication was given by nitrate leaching (43.8±5.63%) in the water and ammonia emission (43.7±4.23%) to the air. The balance of nitrogen at farm level, as a difference between input and output, identified an average surplus of 539±256 kg N ha^{-1}.

3.2.4. Energy use

Farms from this study needed on average 5.97±1.32 MJ from non-renewable energy sources to produce 1 kg of FPCM. Energy consumption was mainly due to off-farm activities (58%). Main components of energy use impact were: production and transportation of concentrate feeds purchased off farm (38.9%), energy use on-farm (20.5%) and feed production on-farm (16%). Energy use on-farm was mainly related to milking, milk cooling, manure handling, feed mixing and distributing. Non-renewable energy use was similar to the results reported by Thomassen et al. (2008) for conventional dairy farms. In a German study, Haas et al. (2001) obtained lower energy consumptions both in intensive and in extensive dairy farms.

3.2.5. Land use

Sample farms occupied 1.51±0.25 m^2 of land to produce 1 kg of FPCM, on average. Land use was quite equally split between on-farm and off-farm activities. Land use impact components were: land use for feed production at the farm (40.0%), land use for production of purchased feeds (34.9% concentrate and 18.8% forages) and land use for energy production off farm (6.4%).
Land use was between the values reported by Thomassen et al. (2008) for conventional and organic Dutch dairy farms.

3.3. Farming strategies and environmental performances

3.3.1. Definition of farming strategies

The results of the factor analysis are showed in table 13.

Tab. 13 - Factor pattern and communalities (h^2) (bold values ≥0.58).

	Factor1	Factor2	Factor3	Factor4	Factor5	h^2
Stocking density, LU/ha	**0.89**	-0.29	0.06	0.26	0.12	0.97
Daily milk production, kg FPCM/cow day	0.14	**0.94**	-0.09	-0.12	0.10	0.94
Production intensity, t FPCM/ha	**0.93**	-0.07	0.02	0.24	0.18	0.97
Feed intake, kg DM per cow	-0.04	0.34	-0.16	**-0.76**	0.22	0.77
Forage, % of DMI	-0.21	-0.37	**0.62**	0.20	0.34	0.71
Maize silage, kg of DMI	-0.04	-0.04	**0.78**	-0.20	0.42	0.83
Concentrate feeds, kg of DMI	-0.14	0.04	**-0.76**	-0.19	0.36	0.76
Feed self-sufficiency, %	**-0.77**	0.13	0.37	0.02	-0.04	0.75
Dairy efficiency, kg milk/kg DMI	0.21	**0.93**	-0.04	0.18	-0.01	0.94
N efficiency, %	0.03	**0.90**	-0.04	0.28	0.10	0.89
P efficiency, %	0.15	**0.81**	0.19	0.37	-0.03	0.86
Maize silage land on farm, % lowland	0.23	0.24	0.24	**-0.58**	0.32	0.60
Grass land on farm, % lowland	-0.25	-0.30	**-0.75**	0.13	0.18	0.76
Input N from purchased feeds, kg/ha	**0.98**	-0.10	-0.02	0.02	0.08	0.98
N balance, kg/ha	**0.97**	-0.05	0.09	-0.12	-0.05	0.97
P balance, kg/ha	**0.94**	-0.12	-0.15	-0.12	0.04	0.93
N farm efficiency, %	-0.37	-0.08	-0.20	0.48	**0.71**	0.91

The model clearly separated the variation into factors: all the variables kept in the model with an eigenvalue higher than 0.58 appeared in only one of the factors. These variables were used to define the farming strategy corresponding to each factor. The first factor was defined as “Farming intensity”; it was characterized by highly positive loading values in terms of stocking density (LU ha^{-1}), production intensity (t FPCM ha^{-1}) and input of nitrogen from purchased feeds (kg ha^{-1}). Moreover, this factor had highly positive values for both nitrogen and phosphorus balances (kg ha^{-}

^{1}), but a negative value for the percentage of feed self-sufficiency. The second factor was named "Animal efficiency" because it was characterized by highly positive values for daily milk production (kg FPCM cow^{-1}) and dairy efficiency (kg FPCM kg^{-1} DMI); nitrogen and phosphorus efficiencies at animal level (%) were also positive. The third factor was identified as "Maize silage intake"; the highest eigenvalue was for the amount of maize silage in cow ration (kg of DMI); the percentage of forage on DMI was also highly positive, whereas concentrate feeds (kg of DMI) and grass land on farm (% land) were characterized by negative values. Factor 4 was defined as "DMI level"; it showed highly negative value for feed intake (kg DMI cow^{-1}); this factor had also a negative value for maize silage land on farm (% land). The last farming strategy was defined as "Farm nitrogen efficiency"; it was only related with efficiency of nitrogen utilization at farm level (expressed as percentage of N output/N input).

3.3.2. Effect of farming strategies on impact categories

The percentage of variation explained by each farming strategy (factor) for each impact category is shown in Table 14.

Tab. 14 - Variation for impact categories explained by each Factor.

		Factor1	Factor2	Factor3	Factor4	Factor5
		Intensive farming	**Animal efficiency**	**Maize silage intake**	**DMI**	**N farm efficiency**
Climate change		0.24	84.2	1.13	10.5	3.93
	P	0.72	<.0001	0.44	0.02	0.15
Acidification		27.0	66.2	1.64	1.44	3.78
	P	0.004	<.0001	0.45	0.48	0.26
Eutrophication		45.6	23.8	14.2	4.02	12.4
	P	<.0001	0.003	0.02	0.20	0.03
Energy use		3.53	45.5	27.9	21.0	2.19
	P	0.37	0.003	0.02	0.03	0.48
Land use		13.9	26.7	0.03	47.1	12.3
	P	0.02	0.002	0.91	<.0001	0.03

The "Farming intensity" strategy significantly contributed to eutrophication, acidification and land use. "Animal efficiency" significantly influenced all impact categories; in particular its load on climate change variation was very high: 84.2% ($P < 0.001$). All farming strategies, with the exception of "DMI level", had an important impact on eutrophication. The most significant contribution to non-renewable energy use came from the "Animal efficiency" strategy. The highest impact on land use was associated to the "DMI level" factor.

Table 4 shows the effects of the first three farming strategies (factors 1, 2 and 3) on each impact category per kg FPCM, comparing the upper and lower quartiles.
"Farming intensity" did not influence very much climate change potential per kg FPCM and there was no difference between the upper and the lower quartiles for GHG production. However, climate change potential off-farm was significantly higher ($P < 0.05$) for the top class of "Farming intensity" in comparison to the bottom one. The relation between farming intensity and off-farm climate change was also identified by Penati (2009). The upper class of "Farming intensity" had significantly higher stocking density (8.62 LU ha^{-1}) than the bottom class (3.46 LU ha^{-1}; $P < 0.001$), lower level of feed self-sufficiency (75.6 vs. 42.3%; $P < 0.001$) and the consequent requirement of high amount of purchased feeds. Off-farm fractions of acidification, eutrophication, energy use and land use were significantly higher in the upper class characterized by high farming intensification and low feed self-sufficiency. As previously showed, the loading value of feed self-sufficiency was highly negative in factor analysis for the strategy "Farming intensity" (-0.77). Penati (2009) found a similar relation between feed self-sufficiency and off-farm fraction of acidification, eutrophication and land use. Comparable results on energy use and acidification were found by Thomassen et al. (2008), who examined conventional and organic farms. Reduced on-farm land use in the intensive farms was also found by Müller-Lindenlauf et al. (2010), who compared low and high input farms.
The strategy "Farming intensity" highly influenced eutrophication and acidification potentials per kg of FPCM and significant differences were registered between the upper and the lower class of intensification: the more intensified farms had always better performances than the less intensified ones. However, considering the eutrophication risks from a local perspective, the results were completely different: nitrogen balance per hectare was significantly higher in the upper class of intensification than in the lower one (854 vs. 299 kg N ha^{-1} per year; $P < 0.001$)
The largest difference in terms of climate change was between the top and the bottom groups identified by the strategy, "Animal efficiency"; GHG production as CO_2 eq. per kg FPCM was 17% lower in the upper class than in the lower one. The upper class had significantly higher milk yield per cow (11429 vs. 8846 kg FPCM; $P < 0.001$) and dairy efficiency (1.45 vs. 1.15 kg FPCM kg^{-1} DMI; $P < 0.001$) than the lower class. As production efficiency (units of milk produced per unit of input) increased, climate change impact decreased, as found by Capper et al. (2009). Kristensen et al. (2011) reported that lower emissions of GHG in efficient farms are mainly due to higher conversion of feed to milk which reduced the CH_4 emission, and, to some extent, to the higher milk yield per cow. A difference of 8% of CO_2 eq. per kg of milk was found by Rotz et al. (2010) in a simulated comparison among farms with different milk production levels. The negative relation between climate change potential and milk production level was also highlighted by Penati (2009). Animal efficiency, expressed as feed conversion efficiency, is increased by the use of fewer animals with high genetic value and high milk production. This effect is based on dilution of maintenance energy requirements, where fewer efficient animals are required to produce the same FPCM $unit^{-1}$ land area. Thus CH_4 emitted and urinary N excreted $unit^{-1}$ product are lower (Beukes et al., 2011). Johnson and Johnson (1995) reported that high producing cows usually receive low fiber rations reducing their methane emission per kg of milk. Diets rich in starch that favor propionate production decrease CH_4 production per unit of fermentable organic matter in the rumen. Conversely, roughage based diets favor acetate production and increase CH_4 production per unit of fermentable organic matter.
The farming strategy based on enhanced animal efficiency showed significantly better performances also in terms of non-renewable energy use per kg of FPCM. Farming systems characterized by high dairy efficiency and nitrogen conversion efficiency at animal level needed and bought less feed for each milk unit produced. As, from our data, purchased feeds

(concentrates and forages) loaded for more than 50% on energy use impact, the better feed conversion saves energy.

The upper class of the farming strategy defined as “Maize silage intake” produced significantly higher eutrophication on-farm than the other; nitrogen from artificial fertilizers was significantly different between classes (82.8 vs. 137.3 kg ha^{-1}; $P < 0.05$). On-farm crop production loaded for 50.8% on eutrophication.

The two classes of “Maize silage intake” were significantly different in terms of feed self-sufficiency (47.6 vs. 64.4; $P = 0.01$). The maize silage achieves an higher energy production, in term of Net Energy for Lactation per ha, compared to other forages, that can also explain the difference found for land use on-farm, that was higher in the upper class of this farming strategy.

The lower class of the “DMI level” strategy had lower energy use and land occupation than the upper class. Dairy efficiency, expressed as kg milk kg^{-1} DMI, and nitrogen animal efficiency were higher in the bottom class than in the upper (1.35 vs. 1.25, $P < 0.10$; 28.0 vs. 26.0%, $P < 0.09$ respectively). The animals of the bottom class of “DMI level” produced the same amount of milk (10.18 vs. 10.24 t FPCM cow^{-1}; $P = 0.90$) with lower level of DMI (20.7 vs. 22.5 kg DM cow^{-1} day^{-1}; $P < 0.001$).

The farming strategy identified as “Farm nitrogen efficiency”, did not show any difference in terms of impact categories; in this case nitrogen efficiency seems to derive mainly from management choices than from animal efficiency. The two classes of quartiles differed in terms of: lucerne land on farm on total farm land (18.0 vs. 8.68 %; $P < 0.05$), input of nitrogen as purchased animal (2.12 vs. 0 kg ha^{-1}; $P = 0.06$), nitrogen fixed by crops (58.9 vs. 40.5 kg ha^{-1}; $P < 0.05$); the lower class had higher nitrogen input than the upper one but these inputs (animals and nitrogen fixing crop) did not produce higher environmental impacts.

3.4. Cluster analysis

3.4.1. Cluster description

From the cluster procedure two main clusters of farms were identified (A and B); moreover in each of the two clusters two subgroups of farms were defined (Figure 13).

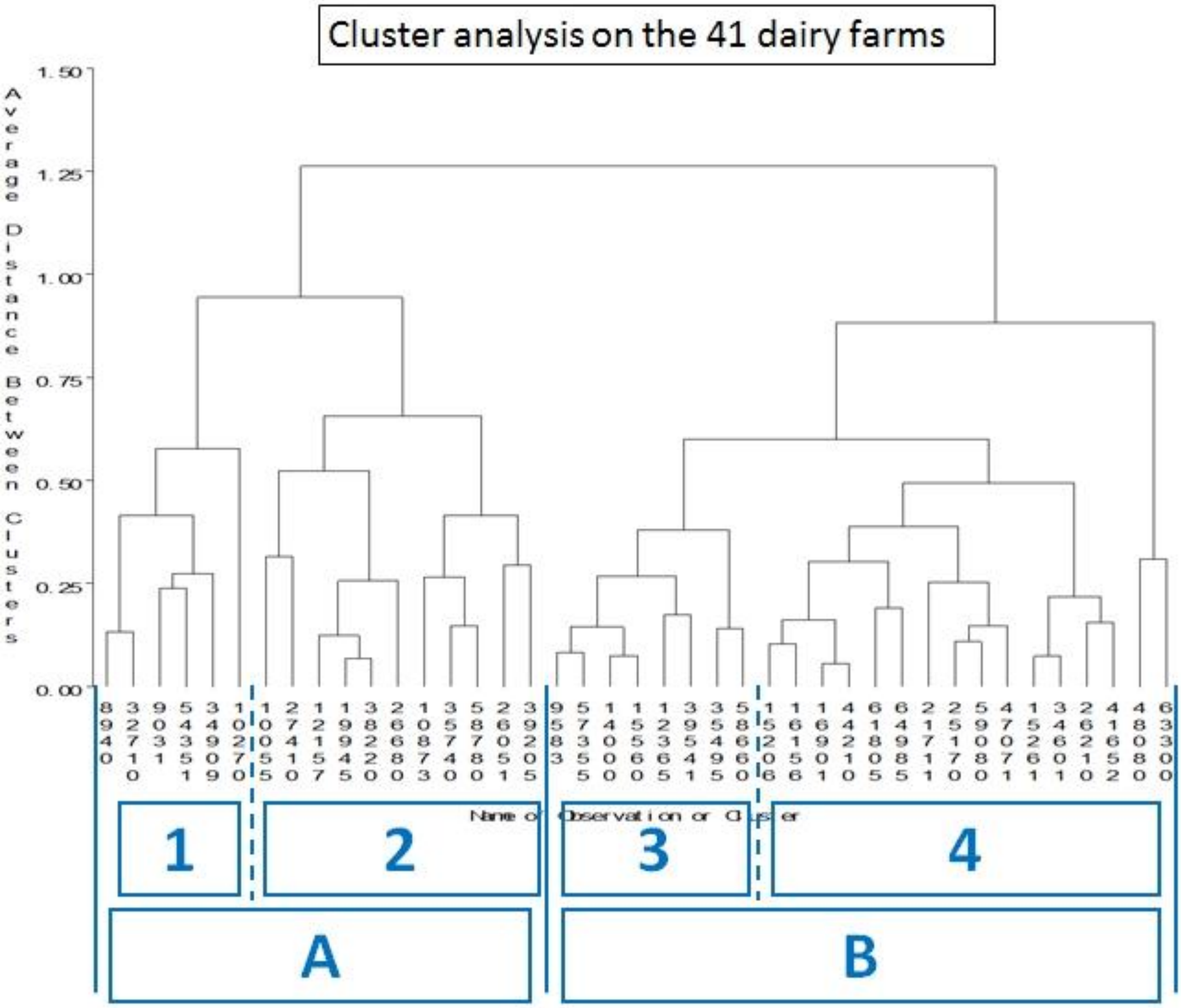


Fig. 3 - The tree procedure average linkage cluster analysis.

In Table 15 are reported the results obtained from the GLM analysis performed on the clusters characteristics.

Tab. 15 - Characteristics of clusters (Least square means).

	Cluster A		Cluster B						
	Cluster 1	Cluster 2	Cluster 3	Cluster 4	SE	P	A vs B	1 vs 2	3 vs 4
n farms	6	11	8	16					
Farm land (ha)	22.8	38.0	57.5	52.9	11.1	0.07	0.01	0.28	0.70
Lucerne on farm land (%)	14.8	14.8	19.1	15.0	5.11	0.87	0.60	1.00	0.45
Permanent grass on farm land (%)	29.8	18.9	1.54	14.8	9.44	0.16	0.04	0.36	0.19
Maize silage on farm land (%)	18.6	31.0	31.3	21.2	7.19	0.30	0.80	0.18	0.19
Livestock Unit (LU)	174.8	224.4	326.0	188.6	50.2	0.07	0.17	0.43	0.01
Stocking density (LU ha^{-1})	9.05	5.84	5.53	3.79	0.87	**<0.001**	**<0.001**	**0.01**	**0.07**
Feed self-sufficiency (%)	31.9	49.8	61.8	73.1	2.42	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Milk production (kg FPCM cow^{-1} day^{-1})	26.8	29.4	29.0	27.6	1.35	0.30	0.86	0.12	0.31
Dairy efficiency (kg FPCM kg^{-1} DMI)	1.27	1.33	1.33	1.29	0.06	0.73	0.95	0.40	0.49
N balance (kg ha^{-1})	852.7	586.6	614.0	349.9	78.8	**<0.001**	**<0.001**	**0.01**	**<0.001**
N feed input (kg N ha^{-1})	1042.4	629.2	564.3	299.9	98.2	**<0.001**	**<0.001**	**<0.001**	**0.02**
N chemic. fert. (kg N ha^{-1})	55.18	113.9	177.8	133.6	21.3	**<0.001**	**<0.001**	**0.03**	0.06
P balance (kg ha^{-1})	125.4	70.4	60.6	31.2	11.8	**<0.001**	**<0.001**	**<0.001**	**0.02**
N farm efficiency (%)	26.0	28.3	26.9	27.2	1.07	0.36	0.92	0.09	0.81
P farm efficiency (%)	28.0	30.9	30.2	29.4	1.45	0.42	0.80	0.12	0.62
Gross margin (€ t^{-1} FPCM)	128.5	206.3	143.2	218.4	20.3	**<0.001**	0.42	**<0.001**	**<0.001**

Cluster A showed a lower but not significant farm land and also the differences regarding the area occupied by the crops had not a statistical importance. The differences for total dimension of the herd (expressed as livestock unit) is not relevant in the statistical model but Cluster A had an higher and significant stocking rate compared to Cluster B, moreover there were statistical differences also between the subgroup of each main cluster in fact Cluster 1 is more intensive than Cluster 2 (with 9.05 and 5.84 LU ha^{-1} respectively) as well as Cluster 3 compared to Cluster 4 (with 5.53 and 3.79 LU ha^{-1} respectively). The level of feed self-sufficiency is strictly connected to stocking rate and that explains why these two variables had the same trend. To an high LU ha^{-1} value was associated a low feed self-sufficiency and vice versa. The production level did not show any relevant difference among the groups as well as the feed conversion rate (dairy efficiency). Both N and P farm balances significant differed between the main clusters and among the sub-groups. Also the farm nutrient balance is generally affected by the stocking rate and consequently by the feed self-sufficiency in fact one of the elements mainly influenced the N surplus is the amount of nitrogen imported in the farms by feed purchased which is statistically different. Despite a higher nitrogen surplus Cluster A showed a lower input of nitrogen per ha from chemical fertilizer, probably farms included in this group paid more attention to chemical nitrogen because their high stocking rate. The results showed that to an high animal load is associated an higher nutrient surplus but on the contrary the farm nutrient efficiency (expressed on the base of nutrient farm balance) did not show any statistical variation among all the groups. From the economical point of view the two main clusters did not present any statistical difference but Cluster 2 had an significant higher gross margin than Cluster 1 as well as Cluster 4 compared to Cluster 3.

3.4.2. Environmental performances

The environmental performances of the two main clusters and of the four subgroups are reported in Table 16.

Tab. 16 - The effect of cluster on total climate change, acidification, eutrophication, energy use and land use per kg FPCM (Least square means).

	Cluster A		Cluster B						
	Cluster 1	Cluster 2	Cluster 3	Cluster 4	SE	P	A *vs* B	1 *vs* 2	3 *vs* 4
n farms	6	11	8	16					
Climate change, kg CO_2eq.	1.43	1.24	1.28	1.31	0.08	0.29	0.55	0.06	0.72
On farm	0.92	0.82	0.90	0.98	0.06	0.07	0.15	0.19	0.23
Off farm	0.49	0.40	0.37	0.33	0.03	**<0.001**	**<0.001**	**0.03**	0.24
Acidification, g SO_2eq.	20.0	18.2	18.7	21.1	1.44	0.17	0.49	0.33	0.12
On farm	14.7	14.1	16.2	19.1	1.19	**<0.001**	**<0.001**	0.68	**0.03**
Off farm	5.15	4.03	2.50	2.01	0.61	**<0.001**	**<0.001**	0.15	0.45
Eutrophication, g PO_4^{3-}eq.	7.9	8.2	8.7	10.1	0.64	**<0.001**	**0.02**	0.76	**0.04**
On farm	4.83	5.78	7.07	8.76	0.56	**<0.001**	**<0.001**	0.18	**0.01**
Off farm	3.07	2.38	1.59	1.35	0.31	**<0.001**	**<0.001**	0.09	0.48
Energy use, MJ	7.13	6.07	5.62	5.65	0.51	0.10	0.03	0.11	0.96
On farm	2.09	1.83	1.85	2.55	0.28	**0.04**	0.31	0.48	**0.03**
Off farm	4.20	3.59	3.50	3.03	0.29	**0.01**	**0.01**	0.10	0.13
Land use, m^2	1.50	1.46	1.46	1.57	0.10	0.62	0.65	0.76	0.31
On farm	0.37	0.57	0.69	0.93	0.07	**<0.001**	**<0.001**	**0.03**	**<0.01**
Off farm	1.12	0.88	0.77	0.64	0.08	**<0.001**	**<0.001**	**0.02**	0.12

Considering the global impacts there were no statistical differences among the main clusters and subgroups, only total eutrophication potential is significant lower in Cluster A and in Cluster 3, this result was not expected because as seen before Cluster A had a statistical higher nutrient surplus which is supposed to be relevant for the impact on eutrophication. As shown by other studies (Haas et al., 2001; Casey and Holden, 2005; Müller-Lindenlauf et al., 2010) the environmental impact estimated in product perspective can differ widely from the impact that the same productive system has on a local scale (i.e. per ha of farm land). For all the impact categories

Cluster A showed a statistical higher impact off-farm, that's probably due to the higher stocking rate of the farms belonging to this group which are for that reason more dependent from external inputs (like concentrate feed, bedding material, energy, etc.). Cluster 1 has an higher off-farm contribution to greenhouse gases emission compared to Cluster 2, the same trend is shown by the land use and the energy use, also in this case the general higher off-farm contribution to global impact in subgroup 1 is linked to the higher animal load. On the contrary Cluster 3 had a significant lower on-farm contribution for all the impact categories compared to Cluster 4. Overall the environmental results showed that the on-farm contribution to GWP, Acidification Potential and Eutrophication Potential is higher than the off-farm quota with values that varied from 65.1% to 74.6%; from 74.1% to 90.5% and from 61.1% to 86.6% respectively. Energy use and land use generally had an higher contribution from the off-farm processes, the values ranged between 54.3% and 66.8% and 52.8% and 74.9%, but the land use on-farm in Cluster 4 is lower compared to land use off-farm.

4. CONCLUSIONS

The study analyzed the environmental impact of milk production, in a cradle-to-farm-gate perspective, in one of the most intensified district of Italian dairy farming.
Average results in terms of climate change, acidification, eutrophication and land use per kg of milk were similar to those obtained by authors from other European countries, even if milk production intensity (t ha^{-1}) of the farms analyzed in the study was higher than in the other studies. Non- renewable energy consumption per kg of milk was quite high, in particular the off-farm fraction. Significant variations were found among different farming strategies in terms of environmental impacts. In particular farming strategies based on high production intensity and high animal efficiency were better able to mitigate environmental impacts per kg milk at farm gate. Increasing production intensity of the farm by increasing stocking density seems to have some advantages on a global scale in terms of acidification and eutrophication potentials per kg of milk produced and did not affect the other impact categories per kg of milk. However, on a local scale the risk of eutrophication associated to this strategy notably increases due to very high nitrogen balance. Another weakness of the intensification strategy is represented by high off-farm fractions of impact categories associated to the high inputs of purchased feeds. As a consequence, the farmer's control on a large part of environmental costs of the process is strongly reduced: environmental impacts of purchased feeds depend on how and where they are produced and they may significantly vary over time. The strategy based on enhanced production efficiency at animal level seems to produce the best results: it allows to reduce GHG production and non-renewable energy use per kg of milk produced. Moreover it does not have negative impact on a local scale in terms of nitrogen balance. High efficiency of feed conversion is mainly associated to balanced and precise feeding and selection of high producing cows. But very high milk yield may have some detrimental effects in terms of animal health, welfare and profitability that cannot be ignored.

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WORK 2: ***ENVIRONMENTAL IMPACT OF DIFFERENT DAIRY FARMING SYSTEMS IN DENMARK, GERMANY AND ITALY.***

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ABSTRACT

The environmental impact of 12 dairy farms in Denmark, Germany and Italy was evaluated using an LCA approach and the most important parameters influencing their environmental sustainability were identified. The farms represent different production methods (organic vs. conventional), summer feeding systems (confinement vs. pasture) and different annual production levels (6275 to 10964 kg ECM cow^{-1}). There was large variability in stocking rate (1.1 to 11.0 LU ha^{-1}) among the farms, which has a major impact on the production per area of farm land (4661 to 61141 kg ECM ha^{-1}), on feed self-sufficiency (28 to 85% of feed dry matter) and on farm surplus of nitrogen (86 to 1001 kg N ha^{-1}). The proportion of grassland on farmland used to produce forages or for pasture varied from 0 to 100%. The lowest global warming potential (GWP), acidification, eutrophication and non-renewable energy use were achieved by the German pasture-based system, followed by the Danish organic dairy system and the very intensive Italian farming system with very similar environmental impact values. There were strong and positive correlations between the four impact categories, and overall the results indicate that improving greenhouse gas emissions would improve the general environmental sustainability of the dairy farm. The land use was lowest in the farms with the highest stoking rate. The organic Danish farms have the lowest impact on biodiversity loss, which in general was positively influenced by the share of grassland in the system. Grassland also had a significant positive effect on GWP, acidification and energy use. The other feature that mainly improved the environmental impact was the feed efficiency of the dairy cows that was positively correlated with GWP, acidification and eutrophication. There was no relation found between the environmental impact and the milk production per cow or the stocking rate at the farm. There was large variation in the relation between on-farm and off-farm contribution between farms and for the different impact categories, showing the importance of a holistic approach and the difficulties in evaluating a farming system both in a product and area-based perspective.

Keywords: LCA, environmental impact, milk, dairy farm, grassland, biodiversity.

1. INTRODUCTION

Production of milk is an example of an agricultural activity that causes environmental side effects, such as emission of greenhouse gases and nutrient enrichment in surface water (Thomassen et al., 2008). In the future the dairy producers will have to meet tighter environmental regulations including limits on greenhouse gas (GHG) emissions and noxious gaseous emissions such as ammonia (NH_3), and stricter nutrient management regulations to control diffuse pollution from nitrate (NO_3) leaching and phosphate (PO_4^{3-}) run-off (O'Brien et al., 2012). Milk production systems vary across Europe, ranging from lowland to highland-based and from extensive to intensive. Increased intensification has exacerbated environmental impacts and the planned removal of the European Union (EU) milk quota system in 2015 (Yan et al., 2011) is expected to result in an increase in milk output and decline in milk price, which presumably will lead to an acceleration of the processes of intensification and specialization (O'Brien et al., 2012). In situations where land availability is a major impediment, producers may decide to adopt alternative production strategies such as confinement systems using a Total Mixed Ration (TMR). In order to be able to devise the best strategy to cope with the new demands, the most efficient and environmentally friendly dairy systems and the parameters affecting these need to be identified. In the last ten years the Life Cycle Assessment (LCA) method has been used in several studies to assess the environmental impact of different milk production systems across Europe, especially comparing organic and conventional system (Cederberg and Mattsson, 2000; de Boer, 2003; Thomassen et al., 2008) or to just evaluate the environmental performance of milk production on a typical dairy farm (Castanheira et al., 2010; Müller-Lindenlauf et al., 2010; O'Brian et al., 2012). Not least when discussing the effect of intensification and change in land use is it important to use methods that go beyond the dairy farm and include the off-farm activities, as illustrated by Kristensen et al. (2011). In a strategic perspective it is important to estimate the environmental impact on several categories and also to address the correlation between these categories and the different management choices of the dairy production systems. Therefore the aim of the present paper was to evaluate the environmental impact of different dairy farming systems across Europe and identify and underline the parameters that most strongly affect the environmental performances estimated in six impact categories of strategic importance for the dairy farmer.

2. MATHERIALS AND METHODS

2.2. Goal and scope definition

2.2.1. Objectives

The goal of this study was to assess the environmental impact of milk production of different farming systems across Europe and to identify the weaknesses and the strengths of the different farm choices and strategies with the purpose of mitigating the environmental pressure. In these study the "standard categories" were complemented by a simplified approach in order to assess biodiversity losses that occur in milk production. Moreover was analyzed the environmental pressure on land unit, considering 1 ha functional unit both on a global and a local scale.

2.2.2. System boundaries

The system under study included the whole life cycle required for the production of raw milk, from the production of inputs to products leaving the farm-gate (cradle to farm gate) , i.e. excluding transport or processing of raw milk. For each dairy farm, a detailed ''cradle-to-farm-gate'' LCA was performed. In particular we considered all the processes related to the farms activity i.e. forages and corps production, energy use, fuel and electricity consumption, manure and livestock management. The external factors (input) like production of fertilizer, production of pesticides, production of fodders and raw materials, production of concentrate feed, production of electricity and fuel, production of litter materials (straw and sawdust) were considered parts of the system. In the farms considered no replacing animals were purchased from the market in the reference year. Related transport associated with the production of purchased inputs (only livestock's feed) was included (Thomassen et al., 2008). All data were referred to year 2010.

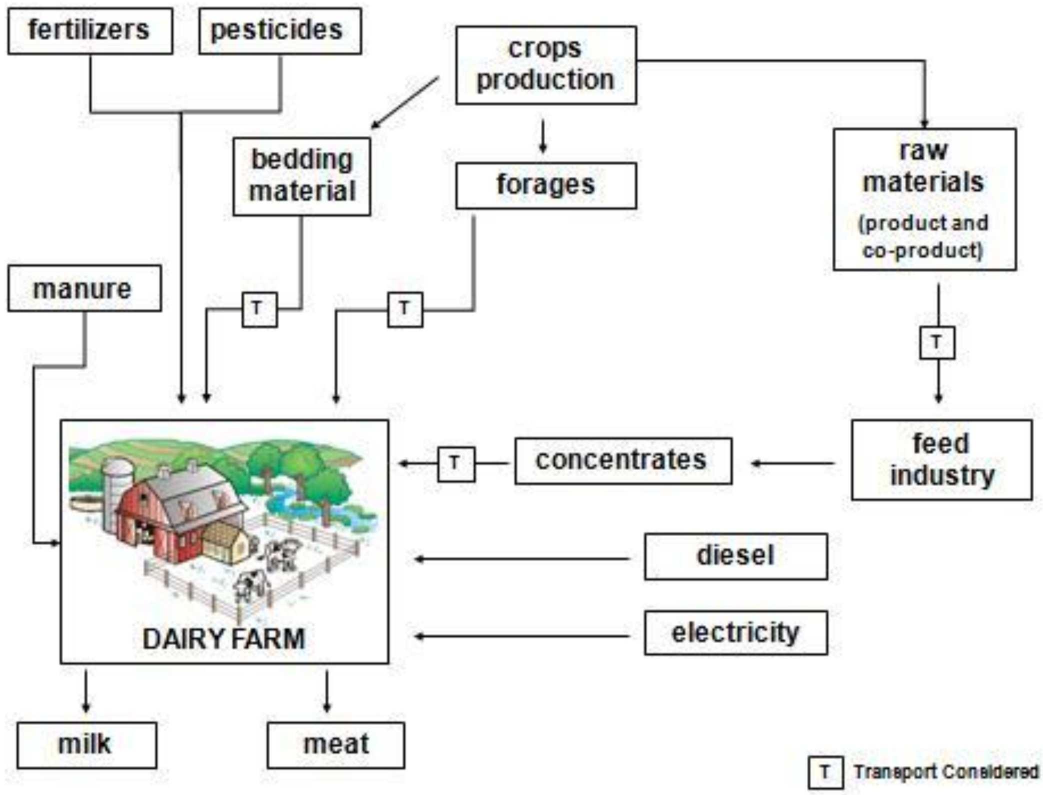

Fig. 2 – System boundaries.

2.2.3. Functional unit.

The functional unit used was 1 kg of Energy Corrected Milk (ECM) (Sjaunja et al., 1990): kg ECM = kg milk*(0.25 + 0.122*Fat% + 0.077*Protein%) delivered to the dairy at the farm gate.
When the environmental impact was addressed to land unit the functional unit considered was 1 ha of farm land and 1 ha of global crop land. For the global crop land were considered for every farm the sum of the on-farm land and total off-farm land used to produce crops for forages, concentrates and bedding materials.

2.2.4. Allocation

In this study a biological allocation developed by IDF (2010), was used. This allocation method is based on the feed energy required to produce the amount of milk and meat at the farm: AF = 1 - 5.7717 x R where AF = allocation factor for milk, R = M_{meat} / M_{milk}, M_{meat} = sum of live weight of all animals sold included bull calves and culled mature animals and M_{milk} = sum of milk sold. Allocation factor for meat is: 1 - AF (allocation factor) to milk.

2.2.5. Impact categories

The environmental impact categories chosen for the study are:

- Global warming potential (kg CO_2 eq. 100-year horizon)
- Acidification (g SO_2 eq.)
- Eutrophication (g PO_4^{3-} eq.)
- Non-renewable energy use (MJ eq.)
- Land use (m^2)
- Biodiversity damage score (DS)

2.3. Inventory analysis

2.3.1. Data sources

Annual data from 12 dairy farms were used: five from Denmark (DK), two from Germany (GER) and five from Italy (IT). The farms were chosen as representative for different cow milk yields and for different stocking rates at the farm, expressed in livestock units (LU) per area of farm land. The five Italian and the five Danish farm where selected following these criteria: DK-3 and IT-3 was representative of the average productive level (kg ECM cow^{-1} $year^{-1}$) and of the average stocking rate (LU ha^{-1}) of the original database, DK-1 and IT-1 had low stocking rate and low productive level, DK-2 and IT-2 had low stocking rate but high productive level, DK-4 and IT-4 had high stocking rate and high productive level and DK-5 and IT-5 had high stocking rate but low productive level. Two of the five Danish (DK-1 and DK-2) farms were organic. The two German farms differed in their summer feeding systems (confinement vs. pasture), while all Italian farms used confinement feeding. The data for the Danish farms were based on intensive registration, while the data used for the GER and IT systems were collected from interviews with the farmers.

2.3.2. On-farm emissions estimation

As proposed by Flysjö et al., (2011) to make the systems as comparable as possible, key figures such as methane emissions from enteric fermentation were calculated using the same model. In the estimation of on-farm emissions we considered the GHG emission (CO_2, CH_4, N_2O) and acidifying substances (NOx and NH_3) and eutrophic compounds (NO_3, PO_4).

2.3.2.1. Greenhouse gases emissions

To quantify the total on-farm air emission were estimated emission from fuel combustion, from enteric fermentation of the herd, emission from manure management (storage and handling including distribution on field), emission that occurs during the application of chemical fertilizers and field urine/fecal deposition during grazing. Emission from livestock respiration are part of a rapidly cycling biological system, where the plant matter consumed was itself created through the conversion of atmospheric CO_2 in organic compounds. Since the emitted and the absorbed quantities are considered to be equivalent, livestock respiration is not considered to be a net source under the Kyoto Protocol (Steinfeld et al., 2006).

CH_4 emissions from enteric fermentation were calculated according to the Tier 2 of IPCC (2006a) guidelines which is based on the dry matter intake of the herd. Knowing the DMI of the herd, then were considered the following factors: the energy content of 1 kg of DM is 18.45 MJ, the methane conversion factor (Y_m: percent of gross energy in feed converted to methane) is 6.5 ±1% and the energy content of methane is 55.65 MJ per kg CH_4 (IPCC, 2006a).

Methane emissions from manure management were estimated using Tier 2 method suggested from IPCC (2006a), in the calculation of the volatile solid excretion rate (VS) we considered that the gross energy content (GE) of the diets (kj/kg DM) was 18.45 MJ for 1 kg of DM (as used before). The CH_4 conversion factors (MCFs) used differs among different storage systems. The

proportion of manure handled with one system or another is based on the number of LU that are confined on litter or on liquid slurry storage system. Methane is also released from manure deposited by animals on pasture and it was estimated with IPCC (2006a) method.
Calculations for N_2O direct emissions from manure storage were based on N herds excretion during confinement period, in according with O'Brien et al. (2012), the quantity of nitrogen excreted from animals was estimated as the difference between the total N intake (calculated as the dietary DM intake and the N content of the diet) and the N output in product (meat, milk). The emission factors used are those proposed by IPCC (2006a) for solid manure and liquid slurry storage systems. Indirect emissions of N_2O from manure storages, that are mainly emissions caused by volatilization of NH_3, were estimated using EF value in according to IPCC (2006a).
Direct and indirect nitrous oxide emissions occurs also at field level after the application of fertilizer in the organic and inorganic forms. N_2O emissions were estimated following the method suggested from IPCC (2006b). Over the amount of nitrogen applied to the soils from synthetic fertilizers an manure (slurry and solid) also the nitrogen inputs from the crop residues were considered.
IPCC's (2006b) methodology was also applied to compute the direct and indirect N_2O emissions that occur during the grazing period considering the amount of urine and dung N deposited by grazing animals.
CO_2 emission related to energy consumption (combustion of fossils fuels and electricity use) were estimated on the base of the amount of diesel used for contract work (l diesel) multiplied by the emission factor 3.31 kg CO_2 eq. l^{-1} diesel (Kristensen et al., 2011).
Carbon sequestration by grassland soils is caused predominately by farm-management changes, for instance the conversion of arable land to grassland (Soussana et al., 2010). The variation in soil carbon stock in the farm land was estimated considering an annual net soil change equal to a gain (sequestration) of 1900 kg of CO_2 per ha of grassland in rotation, to 0 kg of CO_2 per ha of permanent grassland and a loss of 3000 kg of CO_2 per ha of arable land (Kristensen et al., 2011).

2.3.2.2. Other emission on farm level

Nitrogen volatilization from manure storage was estimated multiplying the amount of nitrogen excreted by the emission factors proposed by IPCC (2006a) (30% solid storage, 40% liquid slurry and 28% for pit storage) and in according with Castanheira et al. (2010) they were considered entirely in the form of NH_3.
The volatilization of nitrogen in form of NH_3 and NOx that occur during the application of organic (solid manure, slurry or urine and dung deposited by grazing animals) and mineral fertilizers were estimated using the default emission factors indicated by the Tier 1 in the EEA (2009b) Guidebook that are respectively 0.084 and 0.026 per kg of nitrogen applied.
In nitrogen leaching estimation was assumed that 30% of the N from fertilizer and manure ex storage is lost through in form of NO_3 as proposed by IPCC (2006b). Were also considered like nitrogen input the fractions of N from crop residues and N that is mineralised in association with loss of soil C from soil organic matter as a result of changes to land use or management. To estimate emissions of PO_4^{3-} were considered the amount of phosphorus loss in dissolved form to surface water (run-off) and leaching as proposed by Nemecek and Kägi, (2007) and converted in phosphate (the coefficient of 95/31 is used in order to express the results in kg of PO_4^{3-} instead of P. This method considers the amount of phosphorus excreted by the animals and applied to the field and also the input from chemical fertilizers. The proportion of phosphorus present in solid manure or liquid slurry was calculated on the bases of LU confinement system. These method takes also into account the quota of land that is arable or meadow. Phosphorus loss in particulate form soil erosion to surface water wasn't take into account because lack of data.

In Table 1 are summarized the models, the emission factors and the references used in these study:

Tab. 1 - Equations and emission factors for pollutants estimation on dairy farm level.

Gas	Source	Amount	Emission Factor	Reference
kg CH_4 [a;b]	entheric	CH_4 = kg DMI herd-1 * 18.45 (Gross Energy MJ kg^{-1} DMI) * Ym% / 55.65	Ym% (6.5% ± 1.0%)	IPCC (2006a)
	storages	$CH_4 = VS * B_0 * 0.67 * MCF/100 * MS$	MCF solid storage: 4	IPCC (2006a)
			MCF liquid slurry: 17	
			MCF pit storage: 27	
	pasture	$CH_4 = VS * B_0 * 0.67 * MCF/100 * MS$	MCF pasture: 1.5	IPCC (2006b)
kg N_2O direct [c]	storages	N_2O = Nex (conf.syst.) * MS * EF * 44/28	EF solid storage: 0.005 (0.0027-0.01)	IPCC (2006a)
			EF liquid slurry: 0.005	
			EF pit storage: 0.002	
	field	N_2O = (Nsn + Non + Ncr + Nsom) * EF * 44/28	EF: 0.01 (0.003-0.03)	IPCC (2006b)
	pasture	N_2O = Nex (pasture) * MS * EF * 44/28	EF pasture: 0.02 (0.007-0.06)	
kg N_2O indirect [d]	storages	$N_2O_{G\,=}$ Nvolatilization * EF * 44/28	EF: 0.01 (0.002-0.05)	IPCC (2006a)
	field/ pasture	$N_2O_{(ATDN)}$ = [(Nsn * Frac_GasF) + ((Non + Nprp) * Frac_GasM)]*EF * 44/28	EF: 0.01 (0.002-0.05)	IPCC (2006b)
		$N_2O_{(L)}$ = (Nsn + Non + Ncr + Nsom + Nprp) * Frac_Leach * EF * 44/28	EF: 0.0075 (0.0005-0.025)	
kg NH_3 [e]	storages	Nvolatilization: Nex (conf.syst.) * MS * Frac_GasMS/100 *17/14	Frac_GasMS solid storage: 30 (10-65)	IPCC (2006a)
			Frac_GasMS liquid slurry: 40 (15-45)	
			Frac_GasMS pit storage: 28(10-40)	
	field/ pasture	NH_3 = (Nsn + Non + Nprp) * EF	EF: 0.084 (0.06-0.1)	EAA (2009)
kg NOx	field/ pasture	NOx = (Nsn + Non + Nprp) * EF	EF: 0.026 (0.005-0.104)	EAA (2009)
kg NO_3 [f]	field/ pasture	NO_3 = (Nsn + Non + Ncr + Nsom + Nprp) * Frac_Leach * 62/14	Frac_Leach: 0.3 (0.1-0.8)	IPCC (2006b)
kg PO_4 [g;h;i]	field/ pasture	$P_{gw} = P_{gwl} * F_{gw}$	P_{gwl} arable land: 0.07	Nemecek T. (2007)
			P_{gwl} permanent pasture and meadow: 0.06	
		$P_{ro} = P_{rol} * F_{ro}$	P_{rol} open arable land: 0.175	
			P_{rol} ixtensive permanent pasture and meadow: 0.25	
			P_{rol} extensive permanent pasture and meadow: 0.15	
kg CO_2	soil change	CO_2 = ha * kg $CO_{2\,soilchange}$ ha^{-1}	kg $CO_{2solichange}$ ha^{-1} grassland rotation: + 1900	Nielesen (2003)
			kg $CO_{2solichange}$ ha^{-1} grassland permanent: 0	
			kg $CO_{2solichange}$ ha^{-1} grassland mais and other arable crops: - 3000	
kg CO_2-eq	diesel use	CO_2 eq. = l diesel * EF	EF: 3.31	Kristensen et al. (2011)
	electricity use	CO_2 eq. = kWh * EF	EF: 0.654	

[a] DMI: Dry matter intake; Ym %: CH_4 conversion factor; 55.65: energy content of 1 kg of CH_4

[b] VS: Volatile solid excretion rate eq. 10.24 from IPCC (2006) = [GE*(1-DE/100)+(UE*)]*[(1-Ash)/18.45]; GE: gross energy; B_0: Maximum methane-producing capacity of the manure (Table 10A-4: western Europe = $0.24 m^3 CH_4$/kg VS for dairy cows and Table 10A-5: North America = 0.19 $m^3 CH_4$/kg VS for other cattle); MCF: Manure management methane emission factor derivation (Table 10A-4: annual average temperature = 15°C); MS: Fraction of livestock category's manure handled using a specific manure management system; Ash: 0.08; DE: feed digestibility (Table 10A.1: 70% dairy cows Western Europe and Table 10A.2: 65% growing heifers/steers North America); UE: Urinary energy expressed as fraction of GE (0.04*GE);

[c] Nex: Annual amount of N excreted by the animals; Nsn: annual amount of synthetic fertilizer N applied to soils; Non: annual amount of managed animal manure applied to soil; Ncr: Annual amount of N in crop residues (above-ground and below-ground), eq. 11.7A from IPCC (2006); Nsom: annual amount of N in mineral soils that is mineralised, eq. 11.8 from IPCC (2006); Nmanure import: amount of N imported with manure; Nmanure export: amount of N exported with manure; 44/28: conversion of N_2O-N emission to N_2O emission;

[d] N_2O_G: indirect N_2O emissions due to volatilization of N from manure management; Nvolatilization: amount of manure N that is lost due to volatilization of NH_3 and NOx, all volatilizations are considered in form of NH_3; $N_2O_{(ATDN)}$: indirect N_2O emissions produced from atmospheric deposition of N volatilized from managed soils; Frac_GasF: fraction of synthetic fertilizer N that volatilizes as NH_3 and NOx, kg N volatilized (kg of N applied)$^{-1}$ = 0.1 (0.03 - 0.3); Frac_GasM: fraction of applied organic N fertilizer materials that volatilizes as NH_3 and NOx, kg N volatilized (kg of N applied)$^{-1}$ = 0.2 (0.05 - 0.5); $N_2O_{(L)}$: Annual amount of N_2O produced from leaching and runoff of N additions to managed soils; Frac_Leach: fraction of all N added in managed soils that is lost through leaching and runoff, kg N (kg of N additions)$^{-1}$ = 0.3 (0.1 - 0.8); Nprp: Annual amount of urine and dung N deposited by grazing animals;

[e] Frac_GasMS: percent of managed manure nitrogen that volatilizes as NH3 and NOx in the manure management system, all volatilizations are considered in form of NH_3; 17/14: conversion of NH_3-N emission to NH_3 emission;

[f] 62/14: conversion of NO_3-N emission to NO_3 emission;

[h] P_{gw}: quantity of P leached to ground water; P_{gwl} = average quantity of P leached to ground water for a land use category (kg P ha year^{-1}); F_{gw}: correction factor for fertilization by slurry = 1+0.2/80 * P_2O_{5sl} where P_2O_{5sl} is the quantity of P_2O_5 contained in the slurry;

[i] P_{ro}: quantity of P lost through run-off to rivers; P_{rol} = average quantity of P lost through run-off for a land use category (kg P ha year^{-1}); Fro: correction factor for fertilization with P = 1 + F_{romin} + F_{rosl} + F_{roman}; F_{romin}: correction factor for fertilization by mineral fertilizers = 0.2/80 * P_2O_{5min} where P_2O_{5min} is the quantity of P_2O_5 contained in the mineral fertilizer; F_{rosl}: correction factor for fertilization by slurry = 0.7/80 * P_2O_{5sl} where P_2O_{5sl} is the quantity of P_2O_5 contained in the slurry; F_{roman}: correction factor for fertilization by manure = 0.4/80 * P_2O_{5man} where P_2O_{5man} is the quantity of P_2O_5 contained in the manure;

2.3.3. Off-farm emissions estimation

The estimation of off-farm emissions included the production chain of concentrate feed (from the cultivation of crops to the arrival of the final commercial product to the farm, including processing of raw materials and transport), the production of roughages and bedding material purchased including transportation, the production of chemical fertilizers and pesticides (herbicides and insecticides) but not the related transportation.

Only soy-meal (conventional and organic) and barley grains (conventional and organic) were included as feed concentrates in order to meet the lack of data about commercial feed composition. Data on crop production were taken from Jungbluth et al. (2007) and Nemecek and Kägi (2007). The amounts of soy-meal and barley grains purchased were estimated on the basis of the total N and total DM imported to the farm in commercial feed (excluding roughages).

GHG emissions that occur in the production process of electricity were estimated using the value of 0.654 kg CO_2 eq. kWh^{-1} consumed by the farm (Kristensen et al., 2011).

Emissions related to fertilizer production were estimated using values proposed by Williams et al. (2006). Emissions related to the production of both herbicides and insecticides were estimated using information from Ecoinvent Database (Nemecek and Kägi, 2007).

The estimation of the emissions that occur during the production chain of commercial feed, and also bedding materials, was carried out with the assistance of the Simapro PhD 7.3.3 (Prè Consultants, 2012) software. The transport distances were calculated on the using http://www.viamichelin.it for road trips and http://sea-distances.com/ for the ship transport. The energy consumption for processing 1 ton of livestock feed (about 12% water content) at the factory's gate is considered equal to 53.5 kWh and 140 MJ from heat according to Nielsen (2003).

Only the Italian farms purchased forages from the local market. As described before, emissions on the field level were estimated also including data concerning crops operation (i.e. plowing, harrowing, sowing, harvesting, etc.) taken from Econivent Database (Simapro PhD 7.3.3 - Prè Consultants, 2012).

2.3.4. Biodiversity

Estimation of biodiversity loss was carried out using the method proposed by De Schryver et al. (2010) and also used by Tuomisto et al. (2012) in order to evaluate the biodiversity impacts of contrasting farming systems with alternative land uses in UK. These method estimates the ecosystem damage by using a characterization factors (CFs) that are specific for different land use types with different agricultural practices. The impact on biodiversity is expressed as Damage Score (DS), which describes the relative change in species richness within the occupied area as compared with the baseline. The CF and other data used to assess the biodiversity in this work are shown in Table 3.1.

In our study only the local damage was estimated and referred to 1 m^2 of farms land but also 1 m^2 off-farm land. The local damage score (DS), which is the relative change in species richness for the occupied area. In this study, the CFs for individualistic perspective were used, because De Schryver et al. (2010) only provided the CFs for arable land uses in this approach. The data used to assess the biodiversity in this work are shown in Table 2.

Tab. 2 - Characterization factors (CF) related to type of crop and country of production used for the estimation of biodiversity damage score (DS) for the dairy farming systems.

	DENMARK			GERMANY			ITALY	
CROP*	CF-conv	CF-org	mo	CF-conv	CF-less.int.	mo	CF-conv	mo
Maize silage I [a]	0.79	0.36	12	0.79	0.44	12	0.79	12
Maize silage II [b]							0.79	5
Whole-crop silage	0.79	0.36	5					
Cereal grains-other grains	0.79	0.36	5				0.79	12
Ryegrass/green crops	0.79	0.36	7				0.79	7
Lucerne							0.65	12
Meadow (permanent grassland)	0.65	-0.01	12	0.65	0.36	12	0.65	12
Grassland in rotation	0.65	-0.01	15	0.65	0.36	12		
Beans	0.79	0.36	12					
sugar beet	0.79	0.36	12					

CF: Characterization factor individualistic perspective for each land use:
Intensive arable land: 0.79
Intensive fertile grassland: 0.65
Less intensive arable land: 0.44
Less intensive fertile grassland: 0.36
Organic arable land: 0.36
Organic fertile grassland: -0.01
mo: time of occupation by land use type
[a] sown in spring (April) and harvested in summer (August);
[b] sown in late spring (May) and harvested in late summer (September). This crop follows the harvest of ryegrass that occupies the land during the winter season;
* for off-farm crops are considered intensive arable land and organic arable land (barley and soy) and intensive grassland (grass and lucerne hey);

As proposed by Tuomisto et al. (2012), first the local Damage Score was estimated for 1 m^2 of the farm's land: DS = CF * A * t/12 (where CF is the characterization factor of land use type; and A the area (m^2) occupied by land use type; and the time (in month) of occupation by land use type). Then, to be related to 1 kg of ECM, the DS was multiplied by the value of on-farm land use (m^2 kg^{-1} ECM) carried out in the LCA analysis. The DS linked to the cultivation of off-farm crops was estimated for 1 m^2, differentiating between conventional or organic arable land (soybean and barley) used for concentrate production or grassland used for roughage production and then related to 1 kg of ECM on the basis of off-farm land use (m^2 kg^{-1} ECM). The global value of DS kg^{-1} ECM is the sum of on- and off-farm DS values: DS kg^{-1} ECM = (DS m^{2-1} on-farm)*(m^2 land use on-farm kg^{-1} ECM)+(DS m^{2-1} off-farm)*(m^2 land use off-farm kg^{-1} ECM).

2.4. Impact assessment

Life Cycle Assessment was carried out with the assistance of a commercial LCA software package, SimaPro 7.3.3 PhD (Pré Consultants, 2012). In particular, in the evaluation of GWP, Eutrophication Potential, Acidification Potential and Non-renewable energy use the EPD 1.03 (2008) method was used, updated with IPCC 2007 GWP conversion factors (100 year time horizon) and set the value of CO_2 emission from land transformation to 0. Although emissions due to land use change can be substantial, their quantification and allocation is conceptually and methodologically difficult (Lesschen et al., 2011) and for that reason they were not included in this work. Land occupation was evaluated using Ecological footprint (2009) method. Both methods are present in the software database.

In order quantify the environmental impact in one single value, an analysis with the Stepwise2006 1.03 method was performed (Weidema et al., 2008). This method summarizes the values of all the different impact categories into a single score expressed in monetary units (EUR2003) (Weidema, 2009).

3. RESULTS AND DISCUSSION

3.1. Farm descriptions

The main characteristics of the studied farms are reported in Table 3, divided into herd, land and farm-related results.

Tab. 3 - Characteristics of the studied dairy farms in Denmark (DK), Germany (GER) and Italy (IT).

		DK-1	DK-2	DK-3	DK-4	DK-5	GER-1	GER-2	IT-1	IT-2	IT-3	IT-4	IT-5
HERD													
Cows	No.	168	122	116	127	123	92	36	77	35	98	350	170
Milk production	kg ECM cow^{-1}	6275	7718	8527	10427	7976	10964	6277	10222	6330	9391	10481	7891
Feed intake	kg DM LU^{-1}	6291	5841	6079	7120	6292	6592	4879	6285	6885	6383	6229	5386
Concentrate feed	kg DM LU^{-1}	445	1001	939	1045	3114	2434	159	2770	894	2882	2646	1355
Concentrate feed	kg N LU^{-1}	20	25	70	88	115	112	4	99	50	137	98	55
Concentrate feed	% of DMI herd	27	27	45	40	62	37	3	44	13	45	42	25
Pasture	% of DMI herd	22	25	6	7	0	0	71	0	0	0	0	0
Feed efficiency	kg ECM kg^{-1} DMI cow	0.91	1.18	1.22	1.34	1.19	1.40	1.34	1.31	0.82	1.16	1.40	1.19
N efficiency ex animal	%	18.2	19.7	20.3	22.6	21.9	23.7	18.7	23.3	16.3	21.7	25.6	23.7
N excretion total	kg N LU^{-1}	119	129	134	148	123	141	121	127	145	120	118	101
N excretion pasture	% of herd	22	25	6	7	0	0	71	0	0	0	0	0
LAND													
Area	ha	225.5	162.5	135.7	142.5	74.4	64.0	43.0	58.0	21.4	30.0	60.0	23.0
Maize	% area	2	0	16	32	33	51	0	36	38	53	25	26
Ryegrass+Maize II [a]	% area	0	0	0	0	0	0	0	0	0	23	75	26
Grassland in rotation	% area	47	41	14	44	0	9	90	0	0	0	0	0
Permanent grassland	% area	11	21	12	6	1	41	10	64	34	0	0	30
Lucerne	% area	0	0	0	0	0	0	0	0	8	24	0	17
Total grassland [b]	% area	58	62	26	50	1	49	100	64	42	24	0	48
Land productivity	kg DM ha^{-1}	6374	5178	6065	7169	6831	8563	5261	8847	13286	19478	29071	16387
FARM													
Stocking rate	LU ha^{-1}	1.1	1.1	1.2	1.2	2.1	2.1	1.1	2.2	2.5	5.6	9.8	11.0
Milk intensity	kg ECM ha^{-1}	4661	5523	6722	8695	11863	15692	5255	12690	10343	30686	61141	58325
N fertilizer (organic+chemical)	kg N ha^{-1}	134	141	264	274	290	435	134	340	501	798	1291	1138
N surplus	kg N ha^{-1}	86	89	194	217	224	324	125	177	197	792	1001	498
Feed self-sufficiency (based on DM)	%	92.9	82.9	84.6	85.3	50.5	63.1	96.7	65.1	76.0	54.3	47.5	27.7

[a] sown in late spring (May) and harvested in late summer (September). This crop follows the harvest of ryegrass that occupies the land during the winter season;
[b] sum of grassland in rotation, permanent grassland and lucerne;

3.1.1. Herd

The size of the herd in number of cows varied considerably, from 35 and 36 (IT-2 and GER-1) to 350 (IT-4). The productive levels in kg ECM per year as an average of all cows in the herd varied from around 6300 kg ECM cow^{-1} at farms DK-1, GER-2 and IT-2 to more than 10200 kg ECM cow^{-1}

at farms DK-4, GER-1, IT-1 and IT-4 , compared to the 6300 kg average of EU 15, and an average production in IT, GER and DK of, respectively, 5800, 7000 and 8300 kg milk cow^{-1} in 2011 (Eurostat, 2012). Feed intake as an average of all LU (cows, heifers and calves) at the farm also varied widely in their respective proportions of concentrates and use of pasture. The efficiency of converting dry matter intake (DMI) to milk, which traditionally is both economically and environmentally important, ranged from 0.82 (IT-2) to more than 1.3 kg ECM kg DMI at DK-4, GER-1, GER-2 and IT-4. For ruminants the N efficiency was typically low and in the range 18.2 to 25.6%.

3.1.2. Land

The Danish farms have more land compared to the German and the Italian farms, especially the organic farms DK-1 and DK-2. The differences in land use were both inter-country differences and between farms within a country. At the two organic DK farms, only, respectively, 2.3% and 0% of the land was used to produce maize silage, and neither did GER-2 grow maize for silage, but these three farms instead had the largest share of grassland (in rotation and permanent). Of the Italian farms, IT-1 had the largest share of grassland, all of it permanent. The Danish conventional farms and GER-1 used more land to produce maize silage, and for the Italian farms the share of land used to produce maize silage varied from 36.2% (IT-1) to 100% (IT-4) when also the maize of the second harvest (sown in May following ryegrass) was included. The local climate together with variation in choice of crop partly explains the variation in production from, at around 5200 kg DM ha^{-1} at the low end to more than three times this figure at some of the Italian farms.

3.1.3. Farm

The stocking rate ranged from 1.1 for the Danish organic farms (DK-1 and DK-2) and the German pasture-based farm (GER-2) to 9.8 and 11.0 LU ha^{-1} for the very intensive Italian farms (IT-4 and IT-5). The production intensity, expressed as kg ECM ha^{-1}, was highly variable from less than 5000 kg milk (DK-1) to more than 10 times this at the two farms with highest stocking rate (IT-4 and IT-5). The total amount of nitrogen applied to the soil was lowest in DK-1 and GER-2. For GER-1 the amount was much higher than DK-5 that had the same value of LU ha^{-1} due to a larger application of fertilizer at GER-1 and export of manure for DK-5. For the Italian farms the values ranged from 340 kg N ha^{-1} (IT-1) to 1237 and 1123 kg N ha^{-1} (IT-4 and IT-5). The feed self-sufficiency (expressed as herd DMI) is related to the number of LU ha^{-1}. Generally farms with a low value of LU ha^{-1} are more self-sufficient in feed because they do not need to buy in large amounts of concentrate or forages, and farms DK-1, DK-2, DK-3, DK-4 and GER-2 have the highest values for self-sufficiency, IT-4 and IT-5 the lowest.

3.2. Environmental impact in the product perspective

3.2.1. Impact and process contributions

Table 4 presents the results of the life cycle assessment for the 12 farms. The data show the total environmental impact per kg milk for each category and the contribution from on-farm and off-farm activities, while Figure 2 gives an overall picture of the variability of the results obtained from the environmental assessment of each farms based on a scale from 0 to 1 for each impact category where 0 is the lowest impact.

Tab. 4 - Environmental impact of 12 dairy farms expressed per kg of energy corrected milk (ECM) with the on-farm and off-farm contributions (%).

Impact categories		DK-1	DK-2	DK-3	DK-4	DK-5	GER-1	GER-2	IT-1	IT-2	IT-3	IT-4	IT-5
GWP kg CO_2 eq kg^{-1} ECM	**total**	**1.43**	**1.10**	**1.57**	**1.27**	**1.66**	**1.32**	**0.55**	**1.36**	**1.91**	**1.47**	**1.18**	**1.11**
	on-farm %	95.2	87.5	83.0	85.3	74.7	74.2	93.4	75.2	85.8	73.5	75.1	82.6
	off-farm %	4.8	12.5	17.0	14.7	25.3	25.8	6.6	24.8	14.2	26.5	24.9	17.4
Acidification g SO_2 eq kg^{-1} ECM	**total**	**16.75**	**14.65**	**18.73**	**16.07**	**19.22**	**18.06**	**7.44**	**17.28**	**25.64**	**18.57**	**15.58**	**15.22**
	on-farm %	94.3	89.9	92.4	93.8	83.2	87.2	95.9	85.0	93.6	85.2	85.7	90.4
	off-farm %	5.7	10.1	7.6	6.2	16.8	12.8	4.1	15.0	6.4	14.8	14.3	9.6
Eutrophication g PO_4^{3-} eq kg^{-1} ECM	**total**	**7.56**	**6.37**	**9.17**	**7.50**	**7.78**	**7.69**	**4.61**	**7.06**	**11.12**	**7.70**	**6.22**	**5.85**
	on-farm %	91.4	91.6	94.9	94.6	86.1	89.6	95.1	87.4	95.0	83.2	82.2	84.9
	off-farm %	8.6	8.4	5.1	5.4	13.9	10.4	4.9	12.6	5.0	16.8	17.8	15.1
Non-renewable energy MJ eq kg^{-1} ECM	**total**	**2.87**	**2.55**	**3.08**	**2.96**	**5.29**	**3.71**	**0.92**	**4.09**	**3.73**	**4.12**	**3.37**	**2.40**
	on-farm %	77.1	40.7	50.4	62.5	26.6	26.8	57.4	28.9	49.8	17.7	23.9	29.9
	off-farm %	22.9	59.3	49.6	37.5	73.4	73.2	42.6	71.1	50.2	82.3	76.1	70.1
Land occupation m^2 kg^{-1} ECM	**total**	**3.77**	**3.07**	**2.62**	**2.26**	**3.12**	**1.95**	**3.31**	**2.23**	**1.87**	**2.39**	**1.72**	**1.21**
	on-farm %	89.8	78.6	83.7	82.9	43.2	44.7	94.9	32.0	66.1	26.5	17.3	21.3
	off-farm %	10.2	21.4	16.3	17.1	56.8	55.3	5.1	68.0	33.9	73.5	82.7	78.7
Biodiversity DS kg^{-1} ECM	**total**	**0.54**	**0.45**	**2.23**	**1.93**	**2.73**	**1.48**	**1.14**	**1.70**	**1.40**	**1.86**	**1.35**	**0.93**
	on-farm %	98.3	96.6	84.9	84.3	48.8	42.4	99.7	29.5	64.4	25.7	17.4	20.1
	off-farm %	1.7	3.4	15.1	15.7	51.2	57.6	0.3	70.5	35.6	74.3	82.6	79.9

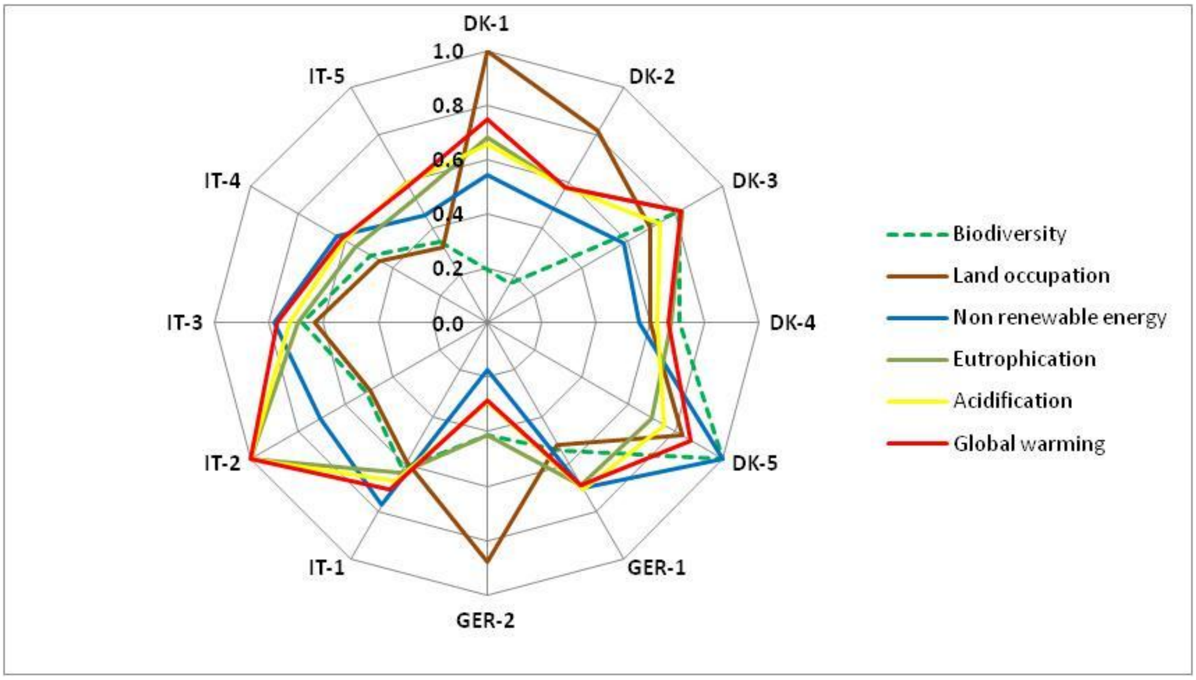

Fig. 2 - Environmental impact from 12 dairy farms expressed in ranked values (0 to 1) within six impact categories.

Based on the energy requirement, the allocation to milk ranged between 76.2% (DK-2) and 90.2% (IT-5) and 91.6% (GER-1) respectively, which means that the latter two farms have a more specialized milk production than the others.

The GWP related to 1 kg of ECM varied from 0.55 (GER-2) to 1.91 kg CO_2 eq. kg^{-1} ECM (IT-2). This agrees with the findings of O'Brien et al. (2012) of a lower environmental impact for a seasonal pastured-based dairy farm than for confinement dairy farms. The lowest-impact Danish farm is the organic DK-2, while in the Italian group the very intensive IT-5 has the lowest greenhouse gas emission per kg ECM.

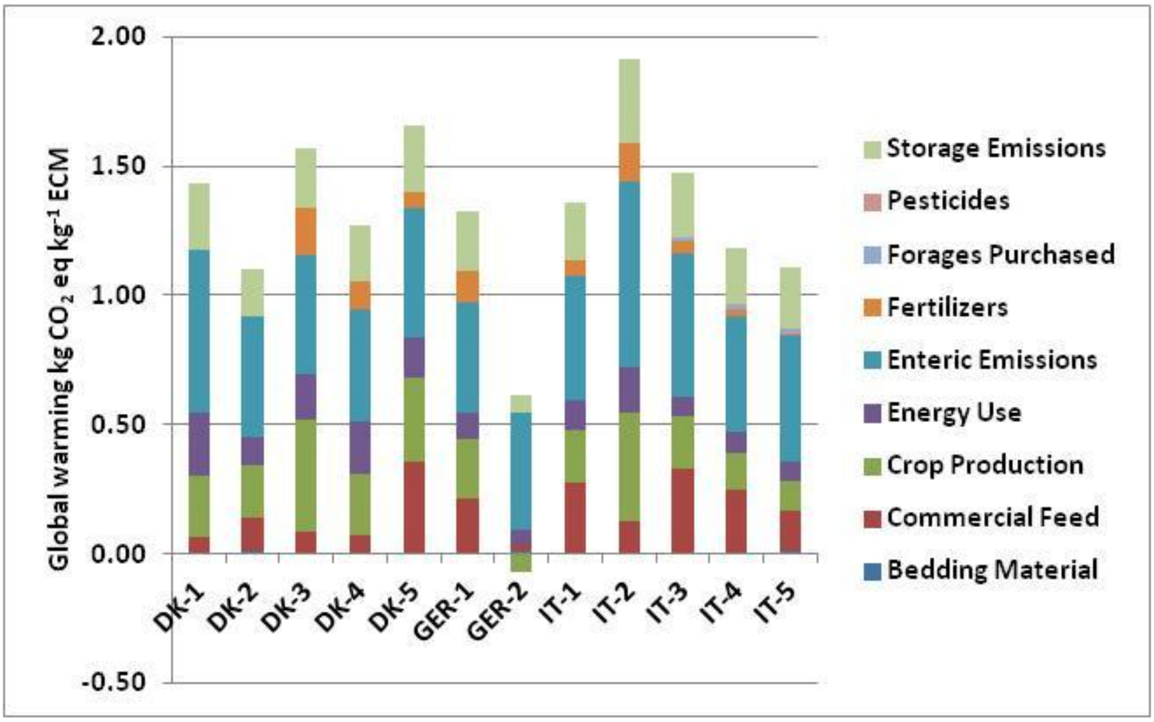

Fig. 3 - Global warming potential at 12 dairy farms (reported in absolute values: kg CO_2 eq. kg^{-1} ECM) and the contribution from different parts of the product chain.

On-farm processes mainly affected the environmental impact. For GWP the quota of on-farm emission vary from 73.5% of IT-3 to 95.2% of DK-1 of total impact on climate change. Enteric emission of methane was the largest contributor to GWP, followed by emissions from manure storage (methane and nitrogen oxides). As known enteric fermentation is the most important source of methane at farm level but also at the global level. Considering all CO_2 eq. emissions, the contribution of enteric methane alone was included between 29.4% of DK-3, 44.5% of IT-5. The total value of both enteric and storage contribution to GWP (mainly in form of CH_4) ranged from 44.1% (DK-3) to 65.9% (IT-5). For GER-2 the contribution of storage emission was lower than at the other farms because the cows here spend around 260 days on pasture. However, the contribution from enteric emissions alone was very high (83.5%). Globally methane is the major contributor to greenhouse gases emissions and its share varied from a minimum of 41.0% in DK-3 to a maximum of 61.1% in IT-5.

On-farm crop production has a strong impact on GWP, and values ranged from 10.5% (IT-5) to 28.0% (DK-3) and 21.9% (IT-2). These values are heavily related to the emissions from fertilizers use (especially N_2O) and also to the variation in soil carbon stock. The negative value for GER-2 (-12.4%) is due to the CO_2 stock in soil exceeding the emissions related to crop production, due to the high proportion of grassland. The other main production section affecting greenhouse gases emissions is the production of commercial feed, where the share for DK-5, IT-1, IT-3 and IT-4 was very high, with respective values of 21.2%, 19.6%, 22.0% and 20.5% compared to DK-1, DK-3, DK-4 and GER-2 where the contribution from commercial feed was at most 6%.

After methane, the other substances which highly contribute to GWP are CO_2 and N_2O, in the average the first one is responsible for 27.3% (± 5.65) and the second one for the 22.0% (± 1.80) greenhouse gases emissions.

The acidification values varied from 7.44 g SO_2 eq. kg^{-1} ECM (GER-2) to 25.64 g SO_2 eq. kg^{-1} ECM (IT-2). The best Danish farm in terms of acidification potential was DK-2 at 14.65 g SO_2 eq. kg^{-1} ECM and for the Italian group IT-5 at 15.22 g SO_2 eq. kg^{-1} ECM. As shown in Figure 4 acidification is strongly influenced by on-farm activities, emissions of storages being particularly important (from 55.2% for GER-2 to 77.9% for IT-5) followed by emissions of crop production (from 11.0% for DK-5 to 40.1% for GER-2). The contributions of both these two compartments to total acidification ranged from 82.3% (DK-5) to 95.3% (GER-2) overall the contribution of on-farm processes to global acidification potential is ranged between 83.2% of DK-5 to 95.9% of GER-2. For GER-2 emissions related to dung and urine deposition from grazing animals are considered part of the crop production emissions.

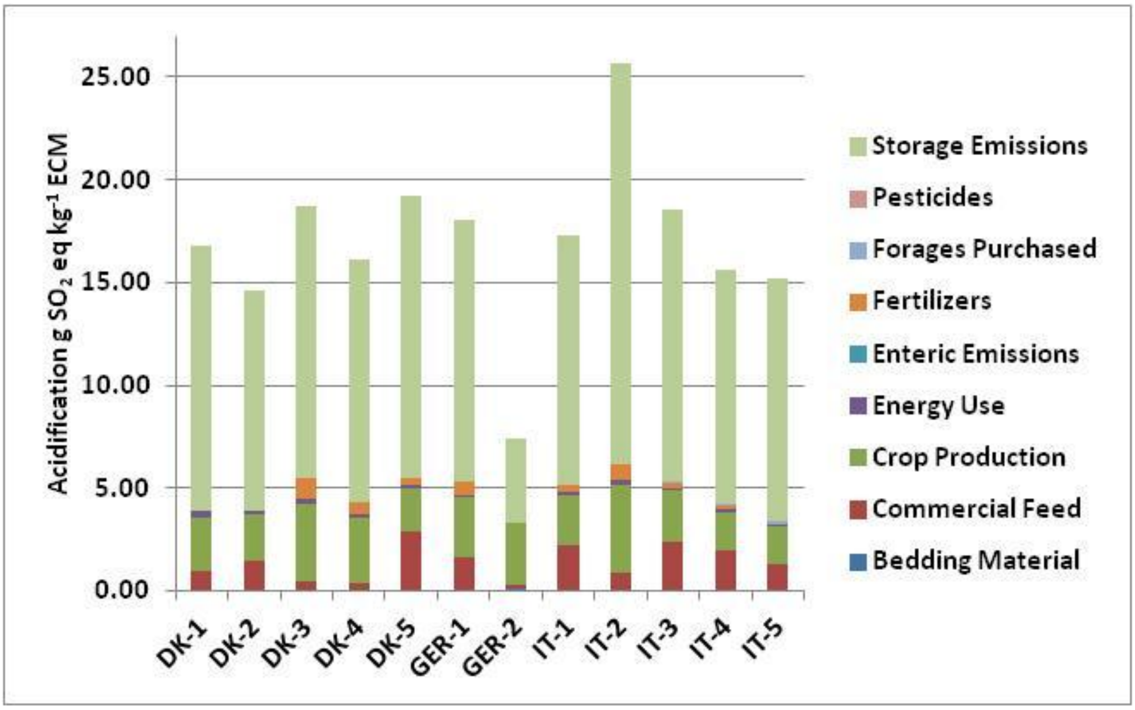

Fig. 4 - Acidification potential at 12 dairy farms (reported in absolute values: g SO_2 eq. kg^{-1} ECM) and the contribution from different parts of the product chain.

Ammonia (mainly from storages and less from crop production) is the substance which strongly contributed to global acidification reaching value of 94.7 % in DK-1 (the lower value is 86.0% shown by DK-5). The lowest and the highest values for eutrophication belong, respectively, to GER-2 and IT-2 at 4.61 and 11.12 g PO_4^{3-} eq. kg^{-1} ECM. Within the Danish and the Italian groups the lowest-impact farms were DK-2 and IT-5 at 6.37 and 5.85 g PO_4^{3-} eq. kg^{-1} ECM, respectively. As shown in Figure 5 on-farm activities are strongly responsible also for the impact on eutrophication, their contribution in ranged between 83.2% of DK-5 to 95.9% of GER-2.

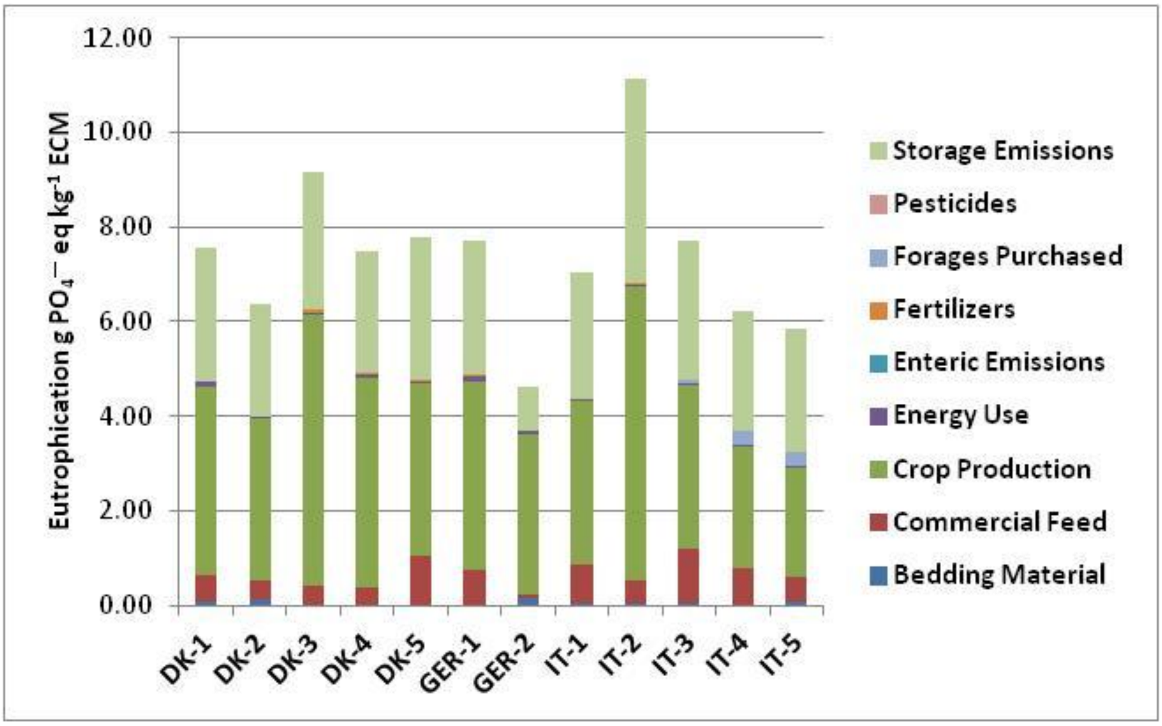

Fig. 5 - Eutrophication potential at 12 dairy farms (reported in absolute values: g PO_4^{3-} eq. kg^{-1} ECM) and the contribution from different parts of the product chain.

Emissions from manure storage (mainly ammonia) and from crop production (especially nitrogen losses due to leaching) contributed from 81.8% (IT-4) to 94.5% (IT-2) to total eutrophication. Nitrate alone showed values which vary from 40.2% to 58.2% of total eutrophication emissions in IT-5 and GER-2 respectively. If only ammonia is considered, its contribution is very important too, and varied from 32.9% of GER-2 to 52.6% of IT-5.

There was no significant contribution of stocking rate to the off-farm impact on GWP, Acidification Potential and Eutrophication Potential, while for Non-renewable energy use, Land use and Biodiversity losses the off-farm impact increased with stocking rate.

As shown in Figure 6, GER-2 had the lowest consumption of non-renewable energy at 0.92 MJ eq. kg^{-1} ECM and DK-5 the largest at 5.29 MJ eq. kg^{-1} ECM. For this category the Danish and the Italian farms with the lowest impacts were DK-2 and IT-5 at 2.55 and 2.40 MJ eq. kg^{-1} ECM, respectively. On-farm activities contributed from 17.7% (IT-3) to 77.1% (DK-1) to non-renewable energy consumption. Production of commercial feed was one of the off-farm activities with the highest energy use and its contribution ranged from 19.0% (DK-4) to 72.3% (IT-3). Also fertilizer production can have a high energy consumption – of the farms that used fertilizers, this activity swallowed from 1.4 % (IT-5) to 29.2 % (DK-3) of the energy used.

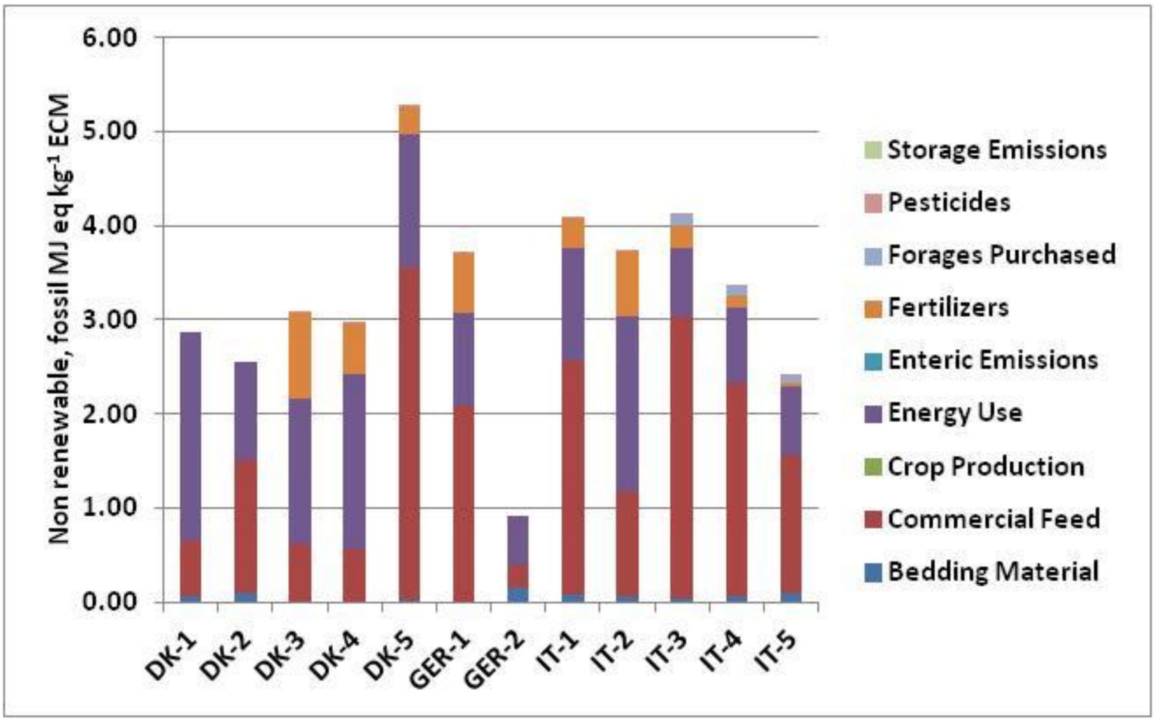

Fig. 6 – Non-renewable energy use at 12 dairy farms (reported in absolute values: g MJ eq. kg^{-1} ECM) and the contribution from different parts of the product chain.

The highest use of land was for GER-2 with 3.31 m^2 kg^{-1} ECM and for the organic farms DK-1 and DK-2 with, respectively, 3.77 and 3.07 m^2 kg^{-1} ECM (Figure 7). This supports the theory that these types of farm generally need more land to produce feed due to their lower crop yields per ha (de Boer, 2003). However the more intensive farms such as DK-5, GER-1 and all the five Italian farms (especially IT-4 and IT-5) had a higher off-farm land use due to a larger amount of imported feed.

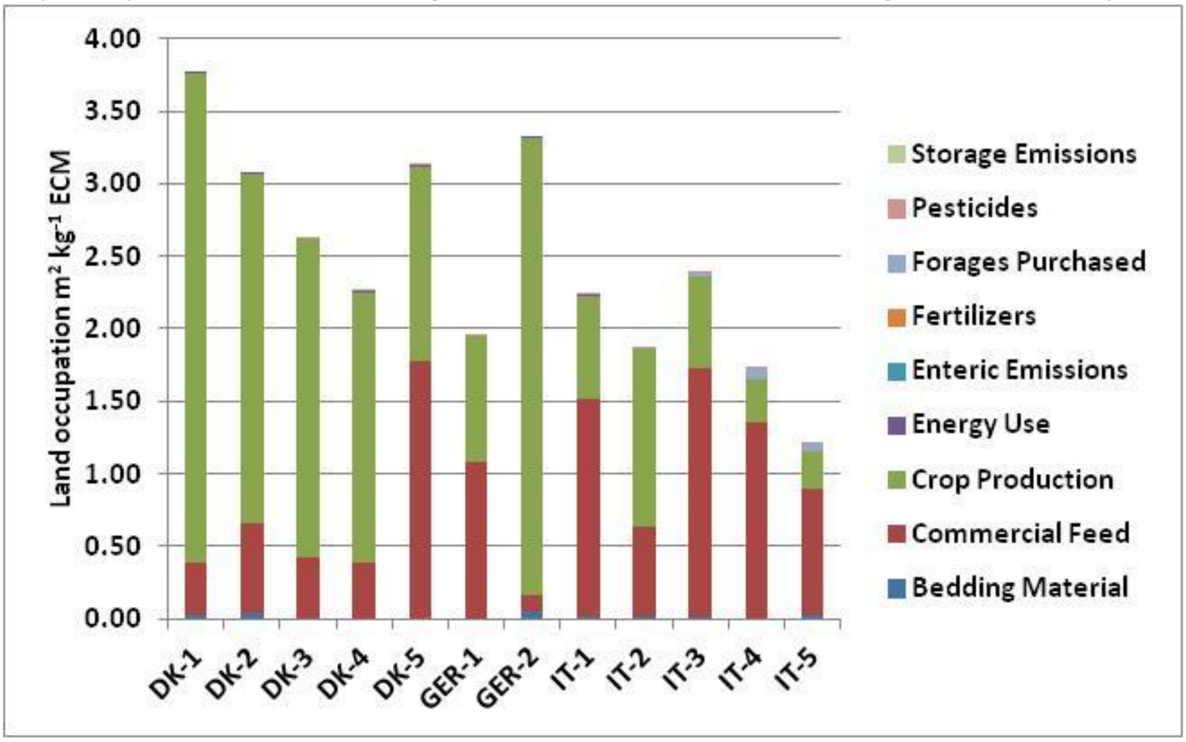

Fig. 7 – Land occupation at 12 dairy farms (reported in absolute values: g MJ eq. kg^{-1} ECM) and the contributionfrom different parts of the product chain.

The last impact category provides an estimation of biodiversity losses related to the different land use types and the values are expressed in Damage Score (DS) kg^{-1} ECM, which explains how the production of 1 kg ECM affects the relative change in species richness.
The farms that had the lowest impact on biodiversity losses were the organic DK-2 and DK-1 at 0.45 and 0.54 DS kg^{-1} ECM, respectively, followed by conventional farm IT-5 at 0.94 DS kg^{-1} ECM. For this category the share of the on- and off-farm impacts was related to the quota of on- and off-farm land use.
Table 5 reports the results from the correlation analysis (SAS, 2009) performed taking all the impact categories into account.

Tab. 5 - Correlation matrix between the impact categories based on data from 12 dairy farms (p-value <0.1 in bulk).

	GWP	Acidification	Eutrophication	Non-renewable energy	Land occupation	Biodiversity DS
Global Warming Potential	1	**0.97**	**0.92**	**0.78**	-0.14	0.46
p-system		**<.0001**	**<.0001**	**0.0027**	0.6723	0.1423
Acidification		1	**0.94**	**0.71**	-0.30	0.33
p-system			**<.0001**	**0.0091**	0.3425	0.2924
Eutrophication			1	**0.53**	-0.13	0.36
p-system				**0.0729**	0.6823	0.2462
Non-renewable energy				1	-0.15	**0.64**
p-system					0.6435	**0.0235**
Land occupation					1	-0.10
p-system						0.9066
Biodiversity DS						1

There are strong and positive correlations between GWP, Acidification Potential, Eutrophication Potential and Non-renewable energy use, while Land use was negatively, albeit non-significantly, related to the four categories. Biodiversity DS showed a significant positive relation to energy use and a non-significant relation to GWP, acidification and eutrophication. Overall the results indicate that improving greenhouse gas emissions would improve the general environmental sustainability of the dairy farm.
The results obtained from the Stepwise analysis show that the EUR2003 single score follows the same trend of GWP, acidification, eutrophication and non-renewable energy use. There is a strong correlation between the single score and these four impact categories (0.80, 0.68, 0.74, 0.60, respectively), confirming the conclusion of the correlation analysis.

3.2.2. Parameters affecting the environmental impact

This paper deals with only 12 farms, each of them based on different production strategies and management efforts. A more general analysis of the relation between the production and the environmental impact therefore has to be handled with care, also due to the geographical bias for some of the expected important farm characteristics like stocking rate, use of fertilizer and crop productivity. Due to the limited number of observations, we only performed one-way correlation analyses between selected farm parameters and the six impact categories as shown in Table 6, combined with a graphic representation of two of the observed significant effects − grassland vs. GWP (Figure 8) and grassland vs. biodiversity DS (Figure 9).

Tab. 6 - Correlation matrix between farm features and the impact categories based on data from 12 dairy farms (*p-value* <0.1 in bulk).

	GWP	Acidification	Eutrophication	Non-renewable energy	Land occupation	Biodiversity DS
Stocking rate (LU ha^{-1})	-0.12	-0.05	-0.29	0.01	**-0.70**	0.18
p-system	0.7063	0.8667	0.3649	0.9773	**0.0112**	0.5802
Milk production (kg ECM cow^{-1})	0.02	0.06	-0.07	0.38	**-0.50**	0.36
p-system	0.9446	0.8433	0.8396	0.2165	**0.0963**	0.2512
Animal N efficiency	-0.20	-0.19	-0.40	0.26	**-0.55**	0.10
p-system	0.5339	0.5587	0.1915	0.4063	**0.0642**	0.7460
N surplus (kg N ha^{-1})	-0.02	0.02	-0.16	0.21	**-0.56**	0.07
p-system	0.9473	0.9513	0.6127	0.5169	**0.0592**	0.8265
N fertilizers (kg N ha^{-1})	-0.08	-0.00	-0.23	0.05	**-0.73**	-0.16
p-system	0.801	0.9909	0.4664	0.8878	**0.0075**	0.6231
Feed self -sufficiency	-0.14	-0.18	0.12	-0.44311	**0.69**	-0.17
p-system	0.6544	0.5775	0.7051	0.1491	**0.0131**	0.6004
Land productivity (DM ha^{-1})	0.06	0.13	-0.06	0.18	**-0.63**	-0.07
p-system	0.8522	0.6828	0.847	0.5741	**0.0288**	0.8396
Production intensity (kg ECM ha^{-1})	-0.14	-0.08	-0.31	0.03	**-0.70**	0.14
p-system	0.6579	0.8128	0.3254	0.9161	**0.0105**	0.6564
Feed efficiency (kg ECM kg^{-1} DMI)	**-0.58**	**-0.57**	**-0.60**	-0.12	-0.23	0.24
p-system	**0.0477**	**0.0528**	**0.0402**	0.7028	0.4736	0.4588
Total grassland (% farm land)	**-0.60**	**-0.53**	-0.38	**-0.72**	0.30	**-0.55**
p-system	**0.0403**	**0.0778**	0.2286	**0.0077**	0.3511	**0.0608**

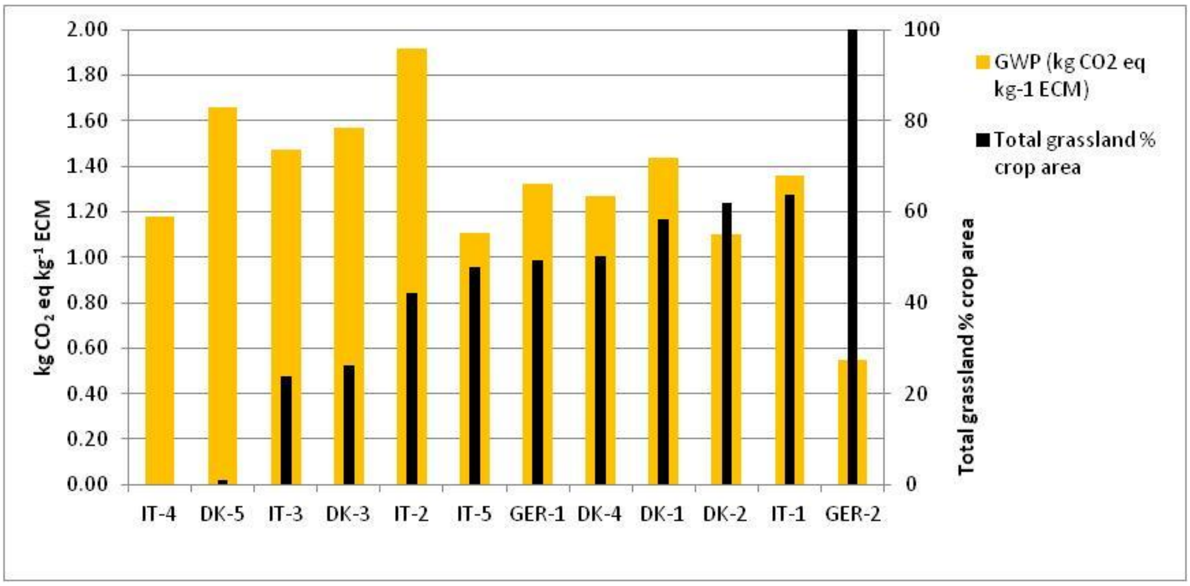

Fig. 8 - Global warming potential at 12 dairy farms (kg CO_2 eq. kg^{-1} ECM) ranked by proportion of grassland on farm cropland (%) .

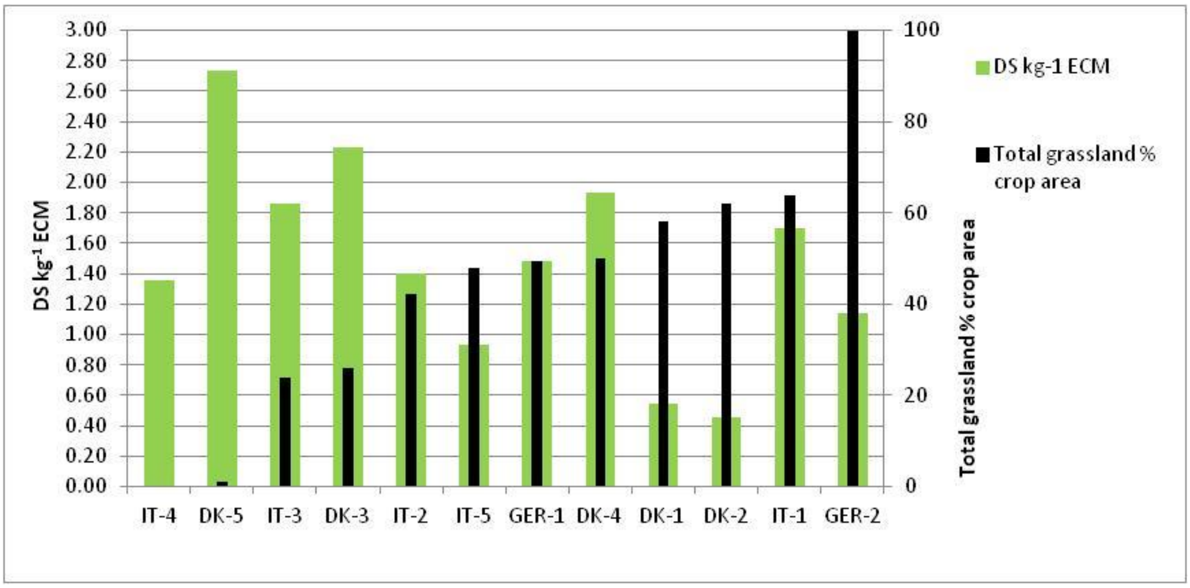

Fig. 9 - Biodiversity damage score at 12 dairy farms (DS kg^{-1} ECM) ranked by proportion of grassland on farm cropland (%).

There were only few significant correlations (p-value <0.1 highlighted in Table 6) between production parameters and either GWP, acidification, eutrophication or Non-renewable energy use, while land use was significantly related to most of the farm parameters, and biodiversity DS was only significantly affected by the proportion of grassland on the farm.

A parameter that affected several impact categories was the feed efficiency and it is important to observe that this feature was negatively correlated with global warming, acidification and eutrophication. This supports the thesis that a better animal efficiency is one of the ways of reducing the environmental impact in milk production (Hermansen and Kristensen, 2011; Opio et al., 2012). Another important parameter that positively influenced GWP, acidification, energy use and biodiversity DS was the share of grassland of the farmed area, where the farms with the largest share of grassland (DK-1, DK-2 and GER-3) had cows on grass during the summer season. The role of grassland in GWP mitigation is probably due to a greater carbon sequestratio. Because of the extensive nature of grasslands, they hold enormous potential to serve as one of the greatest terrestrial sinks for carbon (Opio et al., 2012). Rotz at al. (2010) found a strong decrease in the carbon footprint per kg milk when carbon sequestration in grassland was included Belflower et al. (2012) found a reduction in CO_2 emissions for pasture-based dairy farms compared to confinement farms because fewer field operations are required for tillage, planting, harvesting, and feeding of these crops. This ties in with a greater share of land used to produce grass having a positive effect on energy consumption in our study, with the additional explanation that farms with more grassland are more self-sufficient in feed so they avoid the heavy impact from commercial feed production and transport on total energy consumption. The influence of grassland on decreasing acidification could be related to the lower fertilizer input for this type of crop. Finally grassland plays an important role in reducing biodiversity losses, especially for the organic and pasture-based farms. The positive effect of the proportion of grassland on biodiversity was (as seen in Figure 6) to some extent an indirect effect of the organic farms (DK-1 and DK-2) having the lowest DS and a high proportion of grassland, while the effect within the conventional farms was less clear.

Land use (on- and off-farm) was reduced, in general, when the farming intensity increased (stocking rate, N surplus and use of fertilizer) and – not surprisingly – when crop production on the farmland increased. The first effect was an indirect effect as the farms with the highest intensity were also the farms with the highest import of feed, and the area of land used per DM of imported feed was less than the average area of land used by the farms to produce one kg DM.

A higher production level (kg ECM cow^{-1}) could be expected to reduce GWP per kg of product, as the relative amount of feed for maintenance is reduced, but as shown by Gerber et al. (2011) there is only a minor effect at yield levels below 5000 kg milk cow^{-1}.

We expected to find higher values of acidification and eutrophication for the farms with higher livestock densities (Thomassen et al., 2008; O'Brian et al., 2012), but we found none of these relations to be significant. This result is influenced by the trend of farms IT-4 and IT-5 which, despite a high livestock rate and a consequently high nitrogen surplus per hectare, had a low impact on acidification and eutrophication. The animal nitrogen use efficiency is only weak (p value=0.19) and negatively correlated to eutrophication despite several studies indicating that animal nitrogen use efficiency is a key parameter to improving nitrogen emissions to the environment (Arriaga et al., 2009). Farm GER-2 had the lowest nitrogen use efficiency, but also the lowest impact on eutrophication, which could explain why in this work there was only a slight correlation between this impact category and animal nitrogen use efficiency.

3.3. Environmental impact per ha of farm land and per ha of total crop land used.

The goal of a life cycle analysis is to assess the environmental impact of a product (i.e. 1 kg of milk) which generally is the functional unit of the system under study. In the agricultural sector might be interesting evaluate the environmental impact of a process not expressed only on the final product but also on the land area needed to produce this product. That's for two reason: I) every agricultural system (from forestry to livestock) is based on land use and it influences the land use of a region; II) it makes an environmental pressure on the land (for example nitrate leaching in the soil). From the other side it is important for policy makers to review impacts scaled against the amount of product a system generates rather than the impact per unit area the system occupies, since real reductions in impact need to be balanced against demand for products (Yan et al., 2010).

Only few studies investigated the environmental impact of milk production for an area unit, some researchers have used on-farm (grassland) area (Haas et al., 2001) or on- and off-farm area (Van der Werf et al., 2009; Casey and Holden, 2005; O'Brien et al., 2012) as an FU.

In the present work both the impacts per ha of farm land area and the impact per ha of global crop land area were estimated considering four impact categories: Global Warming Potential, Eutrophication, Acidification, Non Renewable Energy Use.
The impacts per ha on land area were easily estimated changing the functional unit from total amount of ECM produced by the farms with the quota of farm land, in this case all the emissions (on- and off-farm) are loaded on 1 ha of farm land. Also to the global amount of (on-farm + off-farm) land used to produce crops (for concentrate feed, for roughages, for bedding material) were considered, in this scenario all the emissions are spread on the global area and so related to 1 ha of global crop land. In both the scenarios the environmental impacts are allocated between milk and meat taking into account the same allocation values used the previous section. If it could be wired to allocate a process between two products (in this case milk and meat) and address the environmental impact not to these products but to another functional unit (like 1 ha of crop land), it was done because the purpose was to identify the environmental pressure on land unit related to milk production only, avoiding the quota of impact that is linked to meat production.
In Table 7 are shown the data about farm and global land.

Tab. 7 - Land used for crop production.

		DK-1	DK-2	DK-3	DK-4	DK-5	GER-1	GER-2	IT-1	IT-2	IT-3	IT-4	IT-5
	total	**266.1**	**224.3**	**177.9**	**65.3**	**248.5**	**172.4**	**46.8**	**169.5**	**35.4**	**191.5**	**583.1**	**150.9**
Crop land (ha)	on-farm	225.5	162.5	135.7	14.3	74.5	64.0	43.0	58.0	21.4	30.0	60.0	23.0
	off-farm	40.6	61.8	42.2	51.0	174.0	108.4	3.8	111.5	14.0	161.5	523.1	127.9

The GWP expressed on total crop (on- and off-farm) land varies from a minimum 2637.4 kg of CO_2 eq. ha^{-1} in GER-2 to a maximum 11968.4 kg of CO_2 eq. ha^{-1} in IT-2. The average value is 7235.5 (± 2459.2) CO_2 eq. ha^{-1}. These results are comparable with Casey and Holden (2005) who found a greenhouse gases emission for 1 ha of total land ranged between 5918 and 8298 kg CO_2 eq. ha^{-1} performing an LCA on 10 Irish dairy farms, with Van der Werf et al. (2009) who estimated an average emission of 4887 and 6271 kg CO_2 eq. ha^{-1} in 6 organic and in 41 conventional French dairy farms respectively and with Basset-Mens et al. (2009) who observed an environmental burden per land unit ranged between 10453 and 8136 CO_2 eq. ha^{-1} for the more intensive and low input dairy system respectively. O'Brien et al., (2012) observed a reduction in GWP expressed on total farm area in the confinement system compared with the grass-based system (6038 and 9157 kg CO_2 eq. ha^{-1} respectively). If the greenhouse gas emissions are charged only on the farm land the results change widely as can be seen in Figure 10. Clearly the values of CO_2 eq. ha^{-1} incremented in every farms because all emissions are spread on a lower surface but in some cases the values are much more higher than the scenario before: for example farm DK-5 had an impact per ha 3.3 times more, IT-3 and IT-5 6.4 and 6.6 times more respectively and IT-4 9.7 times more, the other farms had a rise ranged between 1.2 and 2.9 times. Moreover the general figure changed especially for IT-3, IT-4 and IT-5 which showed the worst environmental performances even if in the previous perspective their impacts per ha were not the highest. O'Brien et al. (2012) had similar results when the impact was loaded on on-farm land instead total farm land: in the confinement system not only the impact heavily increased but it also showed the worst value compared the grass-based system. Comparing the two scenario of this study there was also a huge difference in data distribution, the variation coefficient was 34.0 % in the first case and 92.0 % in the second case. The results regarding Non-renewable energy use showed the same trend of GWP and when the impacts are loaded on total crop land were obtained values ranged between 4.46 and 23.3 GJ eq. ha^{-1} in GER-2 and IT-2 respectively (with the average value 17.5 ± 5.5 GJ eq. ha^{-1}), similar results were found by Van der Werf et al. (2009) and Basset-Mens et al. (2009). Lower impact was observed by O'Brien et al. (2012).

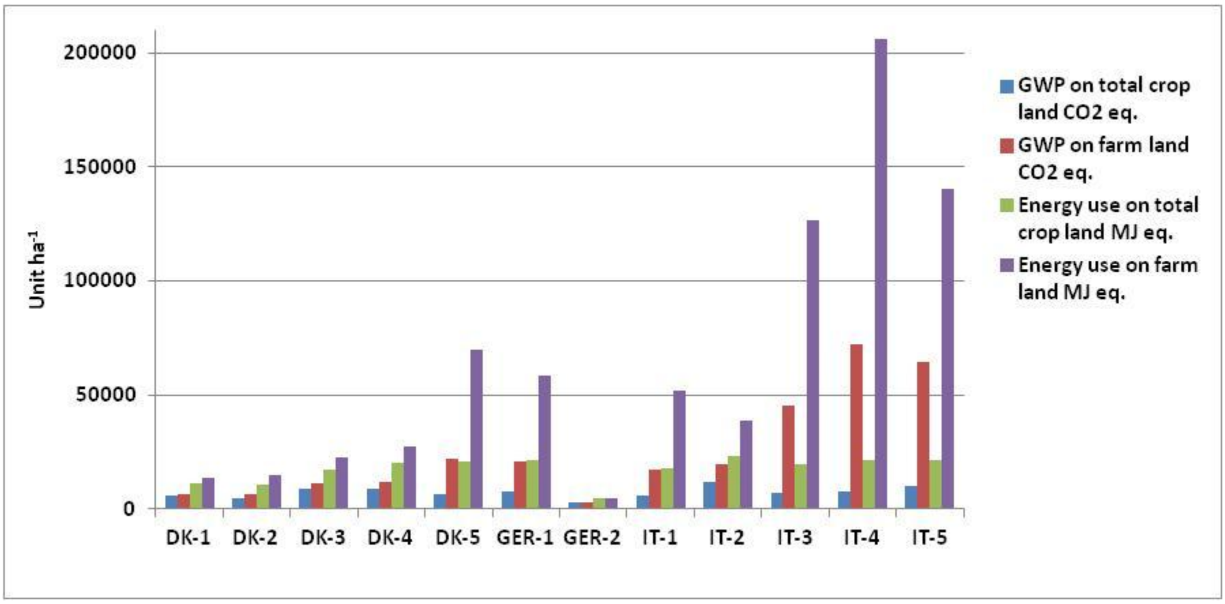

Fig. 10 - GWP and Non-renewable energy use per ha of total crop land and per ha of farm land.

In Figure 11 are represented the results for the Acidification potential an Eutrophication potential. Even if the general trend was clearly the same of the one shown in Figure 10 it was interesting consider also these two impact categories because they are highly linked with the problem of soil and water pollution and nutrients surplus.

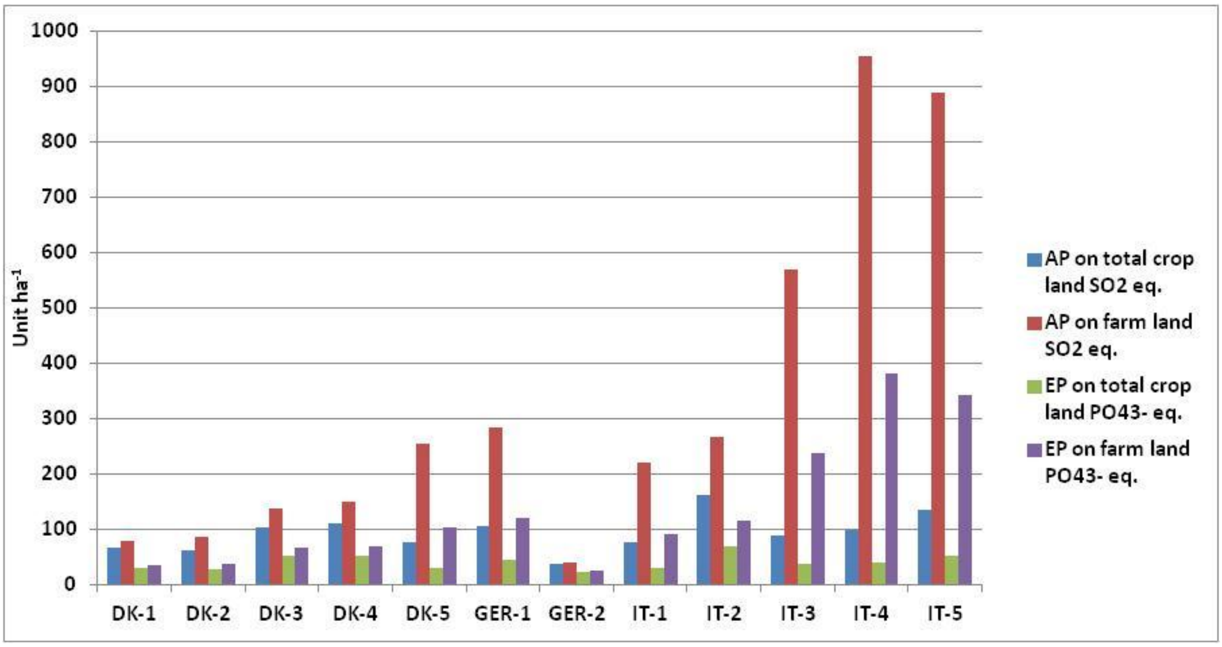

Fig. 11 – Acidification Potential (AP) and Eutrophication Potential (EP) per ha of total crop land and per ha of farm land.

The values of Acidification on total crop land are included ranged between 36.0 and 160.3 kg SO_2 eq. ha^{-1} in GER-2 and IT-2 respectively, the average value is 93.1 (± 33.7) kg SO_2 eq. ha^{-1}, this value is higher than Van der Werf et al. (2009) estimated in his study in fact he found an average acidification potential of 31.0 and 48.1 kg SO_2 eq. ha^{-1} in organic and conventional farms respectively but comparable with what found by Basset-Mens et al. (2009) and O'Brien et al. (2012). The average impact on eutrophication is 40.4 (± 13.7) kg PO_4^{3-} eq. ha^{-1} of total crop land similar to what Van der Werf et al. (2009) calculated for the 41 conventional dairy farms of his work (39.8 kg PO_4^{3-} eq. ha^{-1}) and to O'Brien et al. (2012) who estimated values of eutrophication potential about 40 and 26 kg PO_4^{3-} eq. ha^{-1} for grass-based and confinement system respectively. Basset-Mens et al. (2009) observed a potential impact ranged between 21.4 and 33.0 PO_4^{3-} eq. ha^{-1}. Although in a life cycle approach is more correct consider all the land use worldwide and consequently load the whole impacts on these area it should be taken into account that as seen in this work the greater share of the emissions belonged to on-farm processes, this is particularly relevant for GWP, Acidification Potential and Eutrophication Potential (see Table 3.3) for which were estimated an average on-farm contribution of 82.1% (± 7.63), 89.7% (± 4.33) and 89.7% (± 4.81) respectively.
In order to understand the environmental pressure of the on-farm activities on a local scale might be correct to split the quota of on-farm impact from the total impact and load it to farm land area.
This kind of analysis was done for the Eutrophication Potential which is the impact category that easier can be framed in a local perspective. For every dairy unit the quota of on-farm Eutrophication Potential was calculated considering the on-farm contribution to this impact and then the values obtained were reported on 1 ha of farm land. To make possible a comparison between the on-farm eutrophication potential for 1 ha of farm land and the total eutrophication potential per kg of ECM all the values were ranked between 0 and 1 where one is for both the data series the highest value. The results are summarized in Figure 12.

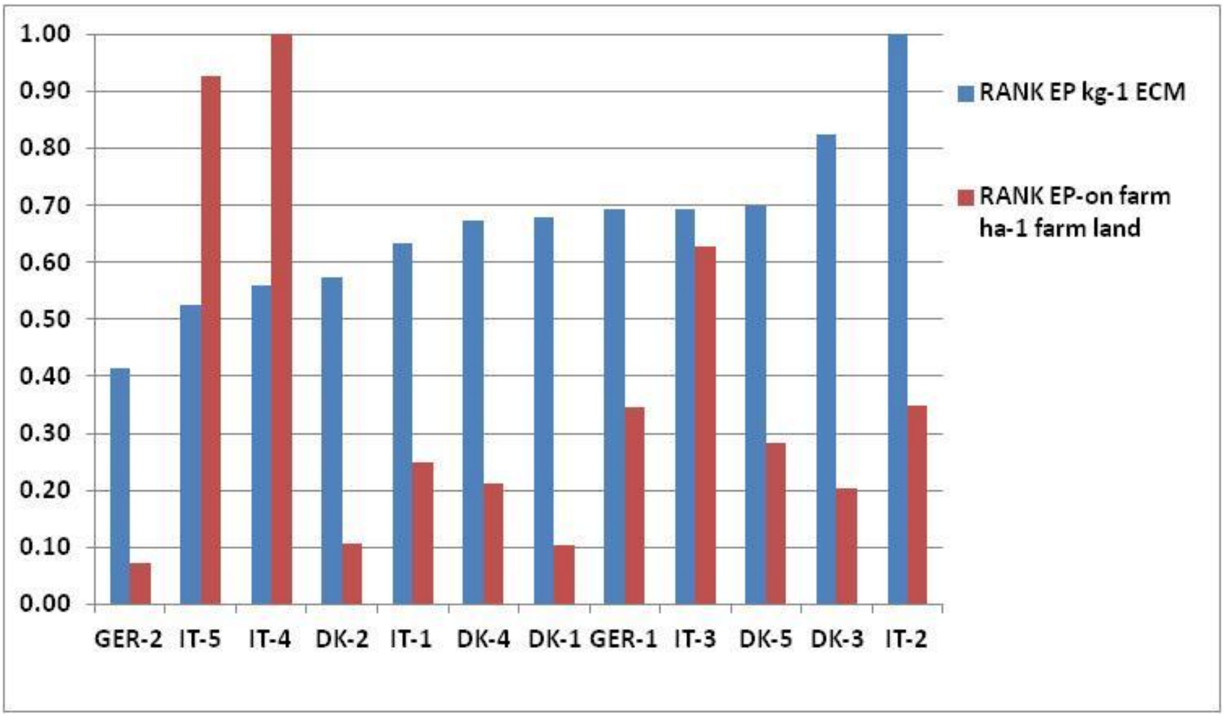

Fig. 12 – Comparison between the on-farm eutrophication potential for 1 ha of farm land and the total eutrophication potential per kg of ECM.

The two scenarios shows completely different trend: DK-3 and IT-2 which had the higher impact per kg of ECM showed a much lower environmental pressure on farm land, but also DK-1 and DK-2 changed their impacts if the eutrophication on 1 ha of farm land is considered. On the contrary IT-4 and IT-5 which had a low eutrophication potential per kg of ECM showed the worst environmental performances against farm land area, IT-3 followed the same trend even if with a low amount of kg PO_4^{3-} eq. ha^{-1} per ha.
The impact per ha of farm land area is strongly correlated with the stocking rate of the dairy farm as can be seen in figure 13.

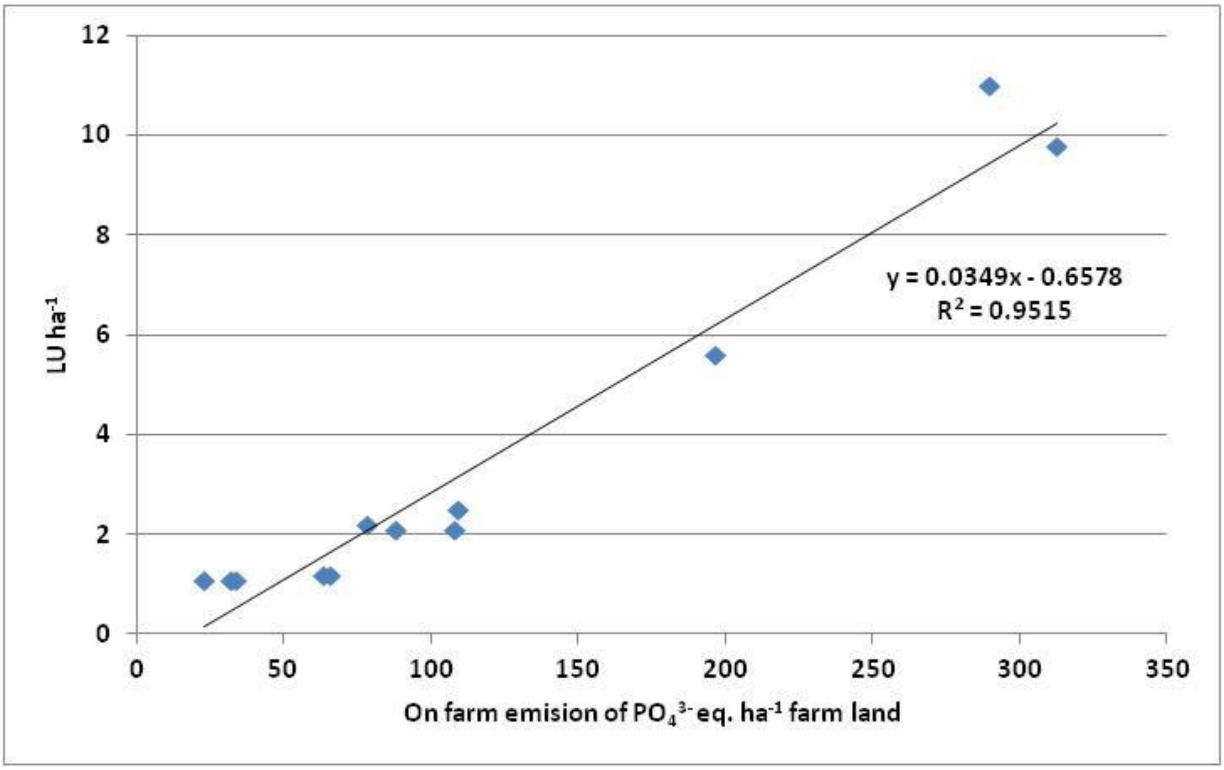

Fig. 13 - Correlation between Eutrophication Potential on farm expressed on 1 ha of farm land and stocking rate (LU ha^{-1})

The Eutrophication Potential was chosen like a “representative” category to describe the two scenarios on the local and the global scale, but the same trend was shown by GWP and Acidification Potential and Non-renewable energy use, even if for this impact category the correlation with the stocking rate is weaker.

4. CONCLUSIONS

The environmental impact of milk production is dependent upon many factors, which is why this analysis did not set out to find the best farming system but to identify the factors influencing the environmental pressure in different farming systems in different countries. The study shows a huge variability in terms of environmental impact within the group of farms analyzed. The parameters that most strongly influenced the environmental impact on 1 kg of ECM were the proportion of grassland in the farming system and the feed efficiency in the herd. There was no relation found between the environmental impact and the milk production per cow or the stocking rate at the farm. In this work we have also shown the result of a first attempt to quantify

the biodiversity losses of producing 1 kg of milk, which was mainly affected by the proportion of grassland in the system.
Most of the impact categories were strongly positively inter-correlated, meaning that improvements to one of them would help improving the general sustainability of the milk chain.
The analysis of the environmental impact on land unit showed that there was a huge difference in results if global crop land or only farm land were considered in impact evaluations. Moreover the environmental load on 1 ha of farm land can be totally unrelated with the impact per unit of product, for that reason a farm with a good environmental performance for 1 kg of ECM can play at the time a negative role on local scale.
Between impact categories and between farms there was large variation in the relationship between on-farm and off-farm contributions, showing the importance of a holistic approach, and the difficulties in evaluating a farming system both in a product and area-based perspective.

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WORK 3: ***CARBON FOOTPRINT OF MILK PRODUCTION IN ITALIAN ALPS THROUGH A SENSITIV APPROACH.***

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ABSTRACT

During the last decades, Italian Alps where characterized by a very high rate of agricultural abandonment that mainly affected small farms. The remaining farms, especially in the dairy sector, showed an evolution trends toward increasing size and intensifying production. The aim of this study was to evaluate the carbon footprint of milk production in mountain area and to investigate the variation of the final results on the base of different methodological assumptions. A "cradle to farm gate" life cycle assessment was performed on a sample of 32 dairy farms and a sensitive analysis was carried out adopting different allocation rules and considering different levels of LUC emission from soybean production. The general farms characteristic showed a small herd dimension (54 lactating cows) but an high stocking rate (3.7±2.0 LU ha^{-1}). The milk production was 6205.7 (±1892) kg FPCM cow^{-1} $year^{-1}$ with an high content of milk solids. The 19.1% (±17.1) of lowland was used for growing maize for silage and only the 39.9% (±41.0) of the herd was transferred for the summer grazing season in the high mountain pasture. The results obtained from the LCA showed an average high GWP (1.60 ±0.27 kg CO_2-eq. kg^{-1} FPCM with IDF allocation and no LUC included). When different allocation method were analyzed the average impact varied from a minimum of 1.57 to a maximum of 1.88 CO_2-eq. kg^{-1} FPCM when system expansion with beef and no allocation were respectively considered. The higher environmental burden of milk production was achieved addressing emission from land transformation to soybean production in Brazil (12.4 and 14.5 kg CO_2-eq. kg^{-1} of soybean and soymeal respectively), in this scenario the contribution of commercial feed to GWP raised 48.3% (±13.0) whereas in the baseline scenario was 19.6% (±6.22). When emissions from LUC where included the results became contradictory.

Key words: Mountain dairy farming, LCA, carbon footprint, sensitive analysis, allocation, LUC.

1. INTRODUCTION

In Italian mountain areas, particularly in the Alps, dairy production is still an important economic activity, strictly connected to the production of typical cheese varieties. In these area the traditional dairy farming is characterized by keeping milking cows and young stock indoors in the lowland most of the year, generally without access to pasture. During summer, however, animals are transferred from the barn in the lowland to pasture in the highland (Penati et al., 2011). Over the decades (1960-2000) the number of dairy cows in the Central Italian Alps (Sondrio Province) decreased by the 40% but the total production of milk incremented by the 14.5%, more intensive

dairy farming systems with higher yields (cows more specialized in milk production), higher stocking rate and consequently lower feed autonomy (with an increasing amount of purchased feed, especially concentrates) replaced the traditional ones, used until nearly half a century (Gusmeroli et. al., 2006). Moreover summer grazing of cows in high altitude pastures (from 600 to 2500 m a.s.l.) has been gradually abandoned (Gios and De Ros, 1991) and more intensive crop systems, like maize, were substituted to the traditional permanent meadows in the valley fields. This situation generates a lot of doubts on environmental sustainability for the mountain dairy farming systems.
Only few studies investigated the environmental impact of milk production in mountain area, some of them focused on farm nutrient balance (Cozzi et al, 2006; Giustini et al., 2007; Penati et al., 2011) and less estimated the CF of milk production (Penati et al., 2010 and Schader et al., 2012).
The increasing concern about greenhouse gas emission from the livestock sector let many European research groups to assess the environmental impact of farm activities in a product perspective with a life cycle approach. LCA is a powerful tool for holistic assessment of environmental impacts of a certain activity but the final results are often affected by a certain degree of uncertainty since relatively arbitrary methodological choices had to be made, and limited data availability necessitated the use of several simplifications and assumptions. For that reason a sensitivity analysis helps to provide an understanding of the relative importance of various input data on the results of a model (Gerber et al. 2010).
The goal of this study was to estimate the carbon footprint of milk production in Italian Alps performing a life cycle assessment. Moreover, a sensitive analysis was done in order to identify the variation in the results adopting different allocation methods and including land use change values for soybean and soy-meal production.

2. MATERIALS AND METHODS

2.1. Data collection

In the present work were involved 32 dairy farms located in the central region of Italian Alps (Sondrio Province). All these farms were members of a cooperative dairy company which produces different kinds of cheese (hard and soft) typical of this mountain area. Data regarding farm activities were collected through personal interview to the farmers, in particular focusing to obtain precise information about herd composition (number of dairy cows and young stock), farm land area and related crop production (area, operations, yield, etc.), housing system, manure management and energy consumption (electricity and diesel). The amount and the costs of fertilizer and pesticide used was asked. Over the amount and the costs also the origin of purchased roughages, of concentrate feed, of purchased bedding material and of purchased replacing animals were included in the interview. Excluded in the study were medicines, washing detergents and minor stable supplies such as disinfectants, salt for cows, etc. (Cederberg and Mattsson, 2000). The farmers provided also data regarding the feeding strategy of the herd (ratio formulation). Information about the amount of milk delivered, milk components and the revenues were provided by the dairy company. The total amount of meat sold from surplus male calves, dairy cows and other animals (i.e. heifers) for slaughter (excluding dead animals) was estimated on the base of the number of animal sold and the standard live weight for each animal category. Information regarding the revenues from meat sold were asked directly to the farmers. Traditionally in the dairy farming system in Alpine Region is used to transfer all or part of the herd in high pasture land for a grazing summer period (generally from June until the end of August), for that reason further questions about this practice (duration of the grazing period, number of grazing animals, amount of milk produced, etc.) were included of the interview. The composition of each concentrate feed was estimated on the base of the raw materials reported on the commercial label with the help of CPM-Dairy Ratio Analyzer Beta V3 software (Cornell-Penn-Miner, 2004). For all the data collected the reference year is 2009.

2.2. System boundaries, functional unit and allocation

Every farm was analyzed in a typical "cradle to farm gate" approach. All the processes related to the on-farm activity (i.e. forages and corps production, energy use, fuel and electricity consumption, manure and livestock management) and related emissions were taken into account. Greenhouse gases emission arising from the off-farm activities like production of fertilizer and pesticides, production of fodders and bedding materials, production of concentrate feed, production of electricity and fuel, breeding of replacing animals were included in the estimation. Related transport associated with the production chain of purchased feed (both commercial feed and roughages) and bedding material was included.

The functional unit chosen was 1 kg fat protein corrected milk (FPCM) leaving the farm gate: FPCM (kg) = (0.337 + 0.116 x % fat + 0.060 x % protein) (Gerber et al. 2010).

The biological allocation method based on the feed energy required to produce the amount of milk and meat at the farm developed by IDF (2010) was used in the estimation of GWP in the baseline scenario ("Scenario 0").

2.3. On-farm emissions estimation

Emission from livestock respiration is not considered to be a net source of greenhouse gases (Steinfeld et al., 2006) and the variation in soil carbon stock at the farm level were not taken into account in the estimation.

Methane emission from enteric fermentations was estimated on the base of dry matter intake of the herd using the equation reported in the Tier 2 of IPCC (2006a) using 6.5% (±1%) of CH_4 conversion factor of gross energy of the diet and 18.45 MJ kg^{-1} DM as the gross energy content per kg of dry matter (IPCC 2006a). The dry matter intake was estimated using the CPM-Dairy Ratio Analyzer Beta V3 software (Cornell-Penn-Miner, 2004) in order to overcome some missing data especially for non-lactating animals and for some farms in which no information regarding animal ingestion were available. The herd dry matter intake during the summer grazing period was estimated using regression equations number 3 proposed by Vazquez and Smith (2000) and data available from Tamburini (2012) regarding the pasture availability, the NDF content of pasture and the quota of legumes in pasture grass. Methane emissions from manure management (including dung deposited by grazing animals) were estimated using Tier 2 method suggested from IPCC (2006a). In the calculation of the volatile solid excretion rate (VS) were considered the same gross energy content of the diet used in the estimation of enteric emission and feed digestibility of 70% for dairy cows and 65% for heifers and calves. The CH_4 conversion factors (MCFs) used varied among different storage systems (4% solid storage, 17% liquid slurry, 27% pit storage, 1.5% pasture). The proportion of manure handled with one system or another is based on the share of LU that are housed in the specific system.

Nitrogen animal excretion (both during housing and grazing period) was estimated considering the nitrogen intake (computed on the base of % CP of diet) and nitrogen output with milk plus the nitrogen stored in the animal bodies during the growing period. Direct and indirect N_2O emissions from storages were estimated as proposed by the Tier 2 method of IPCC (2006a). For the estimation of direct N_2O emitted during manure storages the emission factors used as N_2O-N kg^{-1} N were: 0.005 (0.0027-0.01) for solid storage, 0.005 for liquid slurry, 0.002 for pit storage. Indirect

N_2O was estimated as 0.01 (0.002-0.05) N_2O-N kg^{-1} N volatilized which was quantified as 0.3 (0.1-0.4) for solid storage, 0.4 (0.15-0.45) for liquid slurry and 0.28 (0.1-0.4) for pit storage. To compute the direct and indirect N_2O emissions that occur during manure and synthetic nitrogen application on lowland field and from dung and urine deposited by grazing animals the Tier 1 of IPCC (2006b) methodology was applied, the emission factor (N_2O-N kg^{-1} N) used for direct N_2O were 0.01 (0.003-0.03) for lowland crops and 0.02 (0.007-0.06) for pasture. Indirect N_2O emission occurred after nitrogen volatilization and leaching. Indirect N_2O losses were 0.01 (0.002-0.05) N_2O-N kg^{-1} N volatilized (which was 0.1 (0.03-0.3) from synthetic fertilizer and 0.2 (0.05-0.5) for organic fertilizers) and 0.0075 (0.0005-0.025) N_2O-N kg^{-1} N leached in the soil (which was 0.3 (0.1-0.8) of total nitrogen applied).
CO_2 emission from fuel combustion were estimated on the base of fuel consumption for each farm, the emission factor used is that proposed by Nemecek and Kägi (2007): 3.12 kg of CO_2 per kg of diesel.

2.4. Off-farm emissions estimation

Estimation of greenhouse gases emission which occur during the production of commercial feed (from crop production to the final product delivered to the farm and the transports related to the whole process) was carried out with the assistance of the Simapro PhD 7.3.3 (Prè Consultants, 2012) software mainly using information reported in the Ecoinvent database v.2.0 (2007). The energy consumption for processing 1 ton of livestock feed (about 12% water content) at the factory's gate is considered equal to 53.5 kWh and 140 MJ from heat according to Nielsen (2003b). The emission relative to the production of milk powder was accounted taking the value proposed by Nielsen (2003a). The emissions related to the growing of forages purchased and to the production of bedding material purchased with the transportation included in both these processes were also estimated using the Simapro PhD 7.3.3 (Prè Consultants, 2012) software and the Ecoinvent database v.2.0 (2007). To assess the GHG emission from the production of chemical fertilizers and pesticides (but not the related transportation) was used the Ecoinvent database v.2.0 (2007) except for general N, P and K where it was used Patyk and Reinhardt (1997).
GHG emitted during the production of diesel and electricity were estimated considering the following values: 0.508 kg CO_2-eq. kg^{-1} diesel and 0.58 kg CO_2-eq. kWh^{-1} (Ecoinvent database v.2.0 2007).
Emission which occur in the process of rearing the replacing animals purchased were estimated in a very simplified way considering animals sold at 24 months of age, an average feed ingestion and an average diet composition for each life period, a standard housing condition and manure management.
No land use change (direct or indirect) was taken into account to evaluate the environmental impacts of feed production for the baseline scenario (called “BIO”).

2.5. Impact assessment

For every farm considered in this study the final evaluation of the carbon footprint of 1 kg of milk was carried out in a Excel worksheet. The conversion factors used to transform kg CH_4 and kg N_2O in kg CO_2-eq. were 25 and 298 respectively for an horizon of 100 years (IPCC, 2007). The sum of all the emissions (kg CO_2-eq.) estimated for each process (on and off-farm) divided by the total amount of FPCM sold and allocated for milk production provided the global warming potential for 1 kg of FPCM at the farm gate (expressed as kg CO_2-eq. kg^{-1} FPCM).

2.6. Sensitive analysis

Some of the parameters and assumptions that could strongly influence the final outcomes in a life cycle analysis are the allocation methods (Cederberg and Stadig, 2003; Kristensen et al., 2011; Flysjö et al., 2011, 2012) and the inclusion of direct Land Use Change (dLUC) in model calculation (Flysjö et al., 2012).

In this study a sensitive analysis was performed in order to compare the following assumptions with the baseline scenario ("BIO") with IDF (2010) allocation :

- no allocation between milk and meat ("NO");
- economic allocation based on the proportion of the revenues from the total amount of milk and meat (on the base of animal live weight) sold from each farm ("EURO");
- allocation according to the proportions of milk and meat nitrogen content ("NITROGEN") (Gerber et al., 2010);
- mass allocation according to the proportions of the total amount milk and meat sold ("MASS");

When applying system expansion, the CF of milk equals all emissions from the activities of the dairy farming system (on and off-farm), minus the emissions for producing the same amount of meat (as output from the dairy system) in an alternative meat production systems (Flysjö et al., 2011, 2012). Two different scenario were assumed for the system expansion:

- System expansion "beef" ("Syst.exp.beef"): in which the meat from both the culled dairy cow and the raised dairy calf replaces beef meat produced in Italy. The greenhouse gases emission related to the production of 1 kg of beef meat in Italy was taken from Leip et al. (2010): 12.21 kg CO_2-eq. kg^{-1} meat (total emissions without LULUC, the system boundaries of this study are the farm gates, including slaughtering).
- System expansion "beef-pork-poultry" ("Syst.exp.bpp"): in which the meat from both the culled dairy cow and the raised dairy calf replaces meat from beef, pork and poultry produced in Italy. The greenhouse gases emissions related to the production of 1 kg of pork and poultry meat (live weight at the farm gate) in Italy were also taken from Leip et al. (2010) and were 4.00 and 2.24 kg CO_2-eq. kg^{-1} meat (total emissions without LULUC) respectively. The average GWP for 1 kg of "mixed" meat consumed in Italy was estimated on the proportion of the average national consumption of these three different kind of meet (22.6 % beef, 48.0 % pork; 19.0 % poultry and 10.4 % other which was assumed have the same GWP of poultry (Camera Commercio Milano, 2010).

Despite the great impact on global greenhouse gas emissions and thus on global warming, land use change (LUC) is hardly incorporated into estimations of the GWP in life cycle assessments (Hörtenhuber et al., 2012) because there is no shared consensus on how to include those emissions in CF estimates, as it is very difficult and complex to establish the drivers behind land conversion (Flysjö et al., 2012). However the global demand for soybean for feed has been the major driver of the expansion and conversion of arable land and natural habitats (Opio, et al., 2012). In this study three different values of LUC emissions for soy-meal production in Brazil were considered starting from the assumption that all increase of soybean area was gained at the expense of forest area (Gerber et al., 2010) and all the soybean and soy-meal used in feed production came from Brazil. The three different scenarios obtained from the simulation were compared with the baseline scenario ("Scenario 0": GWP of 1 kg of FPCM no LUC with IDF allocation):

- "Scenario 1": the values estimated from the Ecoinvent database v.2.0 (2007) for CO_2 loss from soil after deforestation for soy production were 1.03 and 0.808 kg CO_2-eq. for 1 kg of soybean (Soybeans, at farm/BR – Ecoinvent process) and soy-meal (Soybean meal, at oil mill/BR U – Ecoinvent process) respectively;

- "Scenario 2": as proposed by Flysjö et al. (2012) the GHG emission from LUC associated with soy production in Brazil was calculated for one year resulting for the total area deforested during a period of twenty years (1990 - 2009) divided by total soybean production in 2009. The total area deforested between 1990 – 2009 for soybean production was estimated assuming an annual average increment of 0.5 Mha (Gerber et al., 2010), the annual GHG emission for one hectare converted from forest land to annual cropland was 37 t CO_2-eq. ha^{-1} as proposed by PAS 2050 (BSI, 2011) and the total amount of soybean produced in 2009 was 57,345,400.00 tonnes (FAOSTAT, 2012). The relative total amount of soy-meal was estimated as 82.6% of soybean (Dalgaard et al., 2008). The LUC emissions obtained were 6.45 and 7.81 kg CO_2-eq. kg^{-1} of soybean and soy-meal respectively considering no allocation between the two co-products (the emissions from land use change were charged to soybean or soy-meal independently);
- "Scenario 3": values taken from Leip et al. (2010) who investigated three different scenarios of land use change in crops production. The third scenario (where converted land is forest) for soy cultivation in Brazil was considered and LUC emissions of 12.4 and 14.5 kg CO_2-eq. kg^{-1} were addressed to soybean and soy-meal respectively;

2.7. Farm nutrient balance

According to Schröder et al. (2003), the farm-gate balance (FGB) of inputs and outputs can be considered a good tool to evaluate the nutrient flows at the farm scale and to improve nutrients management. The inputs and outputs used for calculating the nutrient balances in each farm were defined according to the study by Penati et al. (2011). The raw information about input and output, collected during the interview, were then adjusted using the conversion factors (for N and P). The farm-gate nutrient surplus was calculated as the difference between input and output of nutrient divided by the agricultural area at the farm scale. In this computation no gaseous losses were considered. Moreover it was supposed that the nutrient balance during the summer grazing period in high mountain pasture was in equilibrium with a nutrient surplus equal to zero and for that reason it might be considered not relevant for an environmental point of view as shown in the results of Penati et al. (2011). All the nutrient input and output considered in the farm balance are referred to lowland period and to lowland area.

2.8. Evaluation of economic indicators: IOFC

The income over feed costs (IOFC) is a popular value as it provides a benchmark for a herd or groups of cows reflecting profitability, current feed prices, and actual milk prices (Hutjens, 2007). The IOFC per cow per day was estimated like: [(kg milk cow^{-1} day^{-1}) * milk price (€ kg^{-1} milk)] - feed cost cow^{-1} day^{-1}.

The IOFC implies the estimation of other indicators like feed cost per cow per day that does not reflect milk yield, stage of lactation, or nutrient requirements and feed cost per unit of dry matter that is a useful term when comparing similar regions, breeds, and levels of milk production. IOFC is certainly strongly influenced by the price of milk and feedstuff but also by the capacity of the animal to convert feed ingested in milk. For that reason also the animal feed efficiency was evaluated. Feed efficiency can be defined as unit of milk produced per unit of dry matter intake (DMI) consumed (Hutjens, 2007).

2.9. Statistical analysis

Principal Component Analysis (PCA, proc PRINCOMP, SAS 9.1 (2001)) was used in order to study the relationships among several quantitative variables (including GWP of "BIO" and GWP of "Scenario 3").

3. RESULTS AND DISCUSSION

3.1. Dairy farms description

The average characteristic of the 32 dairy farms are reported in Table 1.

Tab. 1 – Average characteristics of the studied 32 dairy farms.

HERD	Lactating cows (No)	kg FPCM cow^{-1} year^{-1}	DMI lactating cows (kg DMI cow^{-1} day^{-1})	Forages (% DMI lactating cows)	Feed conversion rate (kg FPCM kg^{-1} DMI)
Mean	54	6206	19.4	62.0	1.09
S.D.	61	1892	1.7	9.2	0.18
Min.	14	2265	15.0	47.9	0.72
Max.	300	10041	22.6	80.4	1.44

LAND	Total farm land (ha)	Pasture % total farm land	Permanent grassland % lowland	Maize silage % lowland	N chemical fertilizers (kg ha^{-1} lowland)
Mean	77.5	54.5	79.9	19.1	23.2
S.D.	62.6	34.0	17.2	17.1	34.8
Min.	8.0	0.0	42.2	0.0	0.0
Max.	289.8	96.6	100.0	57.8	121.6

FARM	Stocking rate (LU ha^{-1})	Grazing animals (% total LU)	Farm feed self-sufficiency (%)	N farm balance (kg N ha^{-1} lowland)	N farm efficiency (%)	IOFC (€ cow^{-1} day^{-1})
Mean	3.7	39.5	63.3	241.3	36.3	7.40
S.D.	2.0	41.0	20.9	158.1	15.8	2.40
Min.	1.8	0.0	10.3	39.3	21.1	2.72
Max.	9.8	100.0	95.1	821.8	103.5	12.63

MILK QUALITY	fat %	protein %	casein % total protein	SPC (standard plate count) (CFU ml^{-1})	Spores (No l^{-1})
Mean	4.05	3.60	77.6	19365	737
S.D.	0.19	0.14	0.3	28782	692
Min.	3.67	3.37	76.8	1333	46
Max.	4.44	4.01	78.2	112542	3271

The average number of milking cows is 54.3 (± 61.2), this value is slightly lower compared to the regional average which is 76 dairy cows per herd (Rama, 2012). The average milk production was 6205.7 (±1892.3) kg FPCM cow^{-1} year^{-1} with a huge difference between the maximum and minimum level. In Lombardy Region the average annual production per cow in 2010 was 9125 kg milk (CLAL, 2012). Compared to our results, Penati et al. (2011) found a similar production level in a sample of dairy farms located in the same area, a lower productivity was performed by the a mountain dairy farms studied by Giustini et al. (2007) and Schader et al. (2012). The daily dry matter intake of the milking cows was 19.3 (±1.69) kg DM cow^{-1} day^{-1}, the average quota of forages in the rations was 62.0% (±9.20), the remaining feed intake was covered by concentrate feed. The quota of maize silage included in the rations showed an average value of 23.4% (±15.1) with a minimum value of 0.0% and a maximum of 49.8%. The feed conversion rate was 1.09 (±0.18) kg FPCM kg DMI^{-1}, lower compared to results obtained by Bava et al., (2012) on a group of 22 dairy farms located in the flatland (Po Valley) of North Italy.

The average total farm land was 77.5 (±62.6) ha, but if only the lowland was considered the average is reduced to 26.4 (±27.8) ha. In the farms studied more than half of the total land was occupied by high pasture which could be used for feeding animals only for a limited period of the year (approximately three months of summer grazing) and with a lower yield per hectare compared to grassland located in lowland. The crop system of lowland was characterized by permanent grassland used for the production of forage like hey and hey silage, the arable crops is mainly maize used for silage. The quota of grassland on total lowland is 79.9 % (±17.2) the remaining quota is used to grow maize.

The stocking rate was calculated considering the total number of livestock units divided only by the lowland area because the herds spent the most part of the year (about nine months) kept close in the barns in the valley. The average livestock rate was 3.74 (±1.95) LU ha^{-1}, higher than Penati et al. (2011) and much more than was found in other studies on dairy farms located in mountain area (Cozzi et al., 2006; Giustini et al., 2007; Schader et al., 2012). 24 farms used to transfer all or part of the herd to the high mountain pastures, but of these farms, only 9 let all the animals to graze during the summer season. The farm feed self-sufficiency is expressed as the ratio between the feed DM produced in the farm and the total DM used for animal feeding (excluding the feed produced on farm but sold), it provides an information about the dependence of the farms to buy feed from the market, the average value of the 32 farms was 63.3 % (±20.9) similar to the values found by Penati et al. (2011) and Bava et al. (2012). The average N and P surplus was 241.3 (±158.0) and 55.4 (±41.5) kg ha^{-1} of lowland respectively. The results from the nitrogen balance of this work were similar to Penati et al. (2011) and to Giustini et al. (2007) and also to what was found by Bassanino et al., 2007 and Segato et al., 2009 for Italian intensive dairy farms located in flat land. The average N and P farm efficiency were 36.3% (±15.8) and 27.1% (±17.2) respectively.

The milk quality was very important because all of the milk was delivered for cheese production, the average fat and protein content of the milk was 4.05% (±0.19) and 3.60% (±0.14) respectively, higher than the regional average which in the same year was 3.91% and 3.46% respectively (Rama, 2012). The casein represented the 77.6% (±0.31) of the total milk protein. Generally the milk produced by these farms showed a low microbial cell count (19365 CFU ml^{-1} ±28781).

3.2. Carbon footprint

In this study the average IDF allocation value for milk was 85.0 % (±7.61) and the GWP was 1.60 (±0.27) kg CO_2-eq. kg^{-1} FPCM, that value is high compared with other recent works (Castanheira et al., 2010; Müller-Lindenlauf et al., 2010; Flysjö et al., 2011; Kristensen et al., 2011; O'Brien et al., 2012; Belflower et al., 2012; Mc Geough et al., 2012). Penati et al., (2010), who performed a life cycle analysis on a group of 31 dairy farms located in the same mountain area, estimated an environmental impact of 1.13 kg CO_2-eq. kg^{-1} FPCM, Schader et al. (2012), who assessed the greenhouse gases emission of two organic dairy farms in Swiss mountain area, found 0.89 and 1.08 kg CO_2-eq. kg^{-1} milk in a mixed farm and in a dairy farm respectively. The on-farm contribution to greenhouse gases (GHG) emission varied from 45.6% to 89.2% and the average value was 66.3% (±8.32) of the total impact. In Figure 1 are shown the contribution of the different compartments to GWP, the main role in on-farm emissions is played by storages and enteric emissions which together represented 53.9% (±7.73) of all GHG emissions, this value was higher than Penati (2009) who found an average contribution to greenhouse gases emission from livestock housing and manure storage of 46% but it was less than Schader et al. (2012) who estimated that these two compartments achieved the 74.7 and 76.4% of global CO_2-eq. emission in the dairy farm and in the mixed farm respectively. The off-farm activity which mainly weighted on GWP were the production of commercial feed (19.6% ± 6.22) followed by the process to rear replacing animals bought by the farms during the year (9.23% ± 7.36).

The total methane emissions at farm level (enteric and storages) contribute for the 75.8% (±3.36) to the on-farm impact and for the 50.3% (±7.67) to the global carbon footprint. If only methane from enteric fermentation is considered its contribution to the on-farm impact was 56.0% (±6.96) and to total carbon footprint was 37.0% (±5.69).

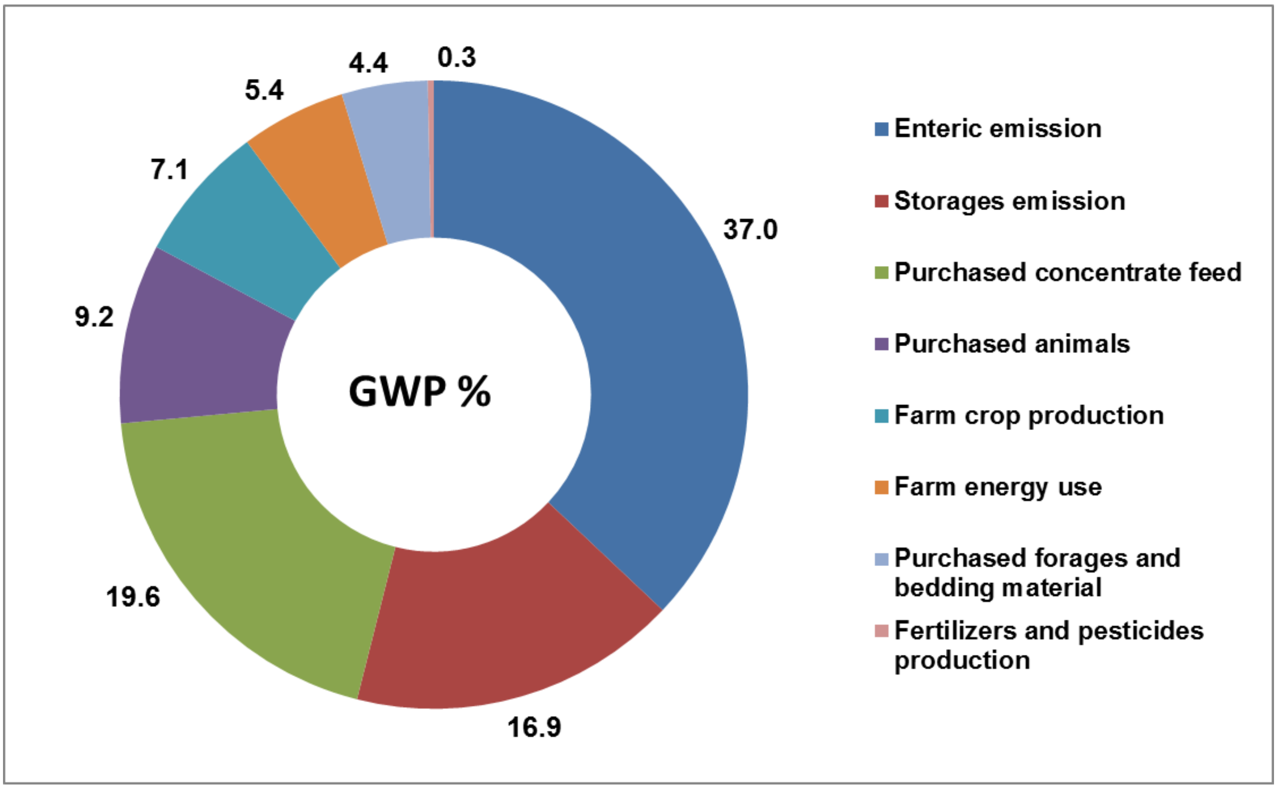

Fig. 1 - Contribution of different compartments to Global Warming Potential.

3.3. Sensitive analysis

A visual representation of how the in an life cycle analysis some methodological choices could have a relevant impact on the final outcomes is shown in Figure 2.

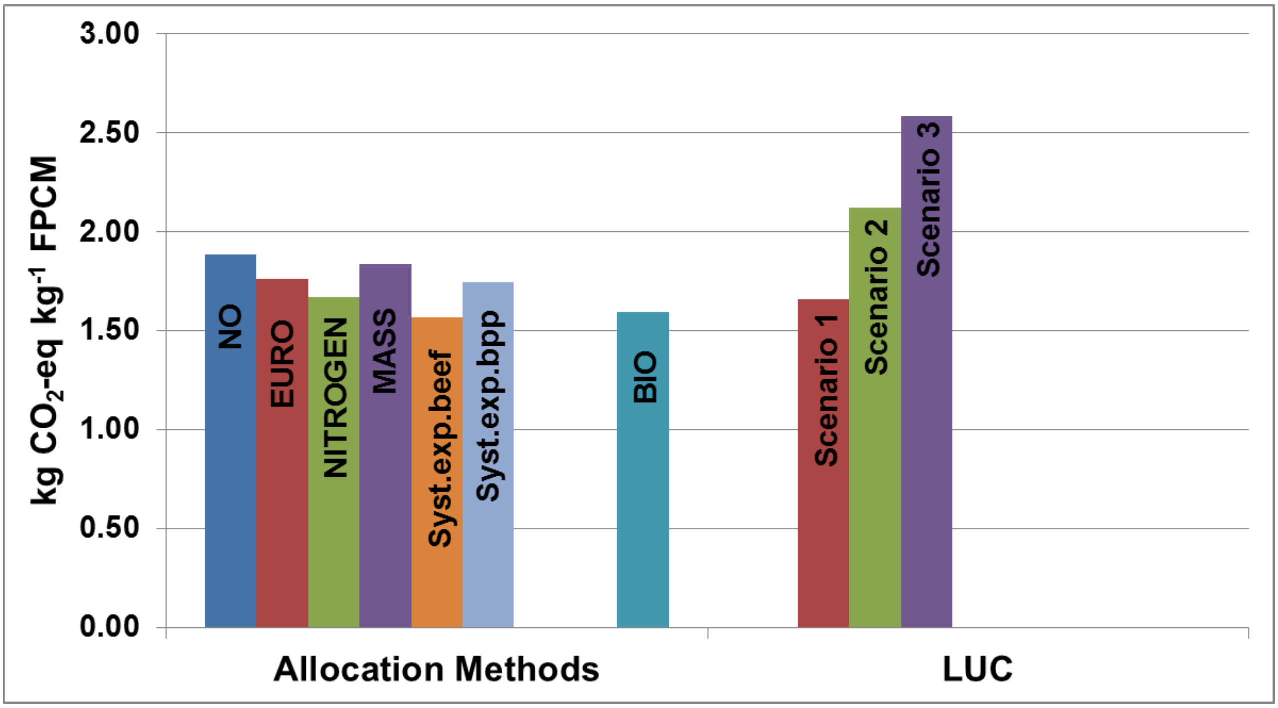

Fig. 2 - Global Warming Potential with different allocation methods and with direct land use change.

3.3.1. *Allocation methodologies*

In Table 2 are reported the results obtained from the sensitivity analysis using different allocation methodologies.

Tab.2 - Sensitive analysis results considering different allocation methods.

	BIO		NO	EURO		NITROGEN		MASS		Syst.exp.beef	Syst.exp.bpp
	Milk %	*CO_2eq kg^{-1} FPCM*	*CO_2eq kg^{-1} FPCM*	*Milk %*	*CO_2eq kg^{-1} FPCM*	*Milk %*	*CO_2eq kg^{-1} FPCM*	*Milk %*	*CO_2eq kg^{-1} FPCM*	*CO_2eq kg^{-1} FPCM*	*CO_2eq kg^{-1} FPCM*
Mean	85.0	1.60	1.88	93.9	1.76	88.8	1.67	97.5	1.84	1.57	1.75
S.D.	7.6	0.27	0.32	4.3	0.26	3.8	0.27	1.2	0.30	0.31	0.30
Min.	59.1	1.24	1.48	79.0	1.39	80.0	1.28	93.4	1.44	1.14	1.33
Max.	97.9	2.52	2.76	98.6	2.37	98.1	2.56	99.6	2.72	2.58	2.68

The lowest average GWP was obtained adopting the IDF allocation (“BIO”) rule which addressed the 85.0% of the total GHG emission to milk whereas the higher impact was clearly achieved when no allocation was considered. Considering the two system expansion scenarios, only the “Syst.exp.beef” showed an impact slightly lower than the baseline scenario (“BIO”), but when different type of meat were taken into account (“Syst.exp.bpp”) the CF of milk was around 9.5% more than “BIO”. Especially for the “beef” scenario a more consistent reduction of the impact was expected. Flysjö et al. (2011), in two groups of Swedish and New Zealanders dairy farms, found a reduction in greenhouse gases emissions of 26.7% and 26.3% respectively considering a system expansion of “beef only” and of 11.6% and 14.1% respectively considering a system expansion “mix” compared to physical allocation scenario (IDF method). Kristensen et al., (2011) did not find a reduction in GWP per kg of milk when system expansion (based 50/50 on the emissions from pig meat and beef meat) was applied compared with biological allocation (IDF method) for both group of conventional and organic Danish dairy farms. Like in our study, in the Danish study (Kristensen et al., 2011) the lower impact for was estimated using the IDF allocation method compared to the other methods (protein mass, economic, system expansion and model A). Also Mc Geough et al. (2012), who performed an LCA on a typical non-grazing dairy production system in Eastern Canada, observed the lower greenhouse gases emission values for 1 kg of milk when IDF allocation “specific” (in which meat-to-milk ratio produced on the simulated farm 26.6:73.4 were used) and IDF allocation “default” (in which meat-to-milk ratio (14.4:85.6) recommended by IDF (2010) were used) where adopted compared to other methods. O’Brien et al., (2012), in a study on different Irish dairy systems, found that the environmental burden of 1 kg of milk increased when an economic allocation was used compared to an allocation factor estimated on the base of energy and protein requirements of the herd.

3.3.2. Direct land use change scenarios

In table 3 are reported the four scenarios obtained when different values for LUC emission were addressed to soybean and soy-meal production.

Tab.3 - Sensitive analysis results addressing different land use change values to soybean and soymeal production.

	Scenario 1			Scenario 2			Scenario 3		
	CO_2-eq kg^{-1} FPCM	diff % [c]	conc.feed [b]	*CO_2-eq kg^{-1} FPCM*	diff % [c]	conc.feed [b]	*CO_2-eq kg^{-1} FPCM*	diff % [c]	conc.feed [b]
Mean	1.66	3.68	22.5	2.12	24.0	38.6	2.58	36.1	48.3
S.D.	0.28	1.94	6.81	0.41	9.37	10.5	0.61	13.1	13.0
Min.	1.29	0.00	6.26	1.30	0.00	17.0	1.30	0.0	17.0
Max.	2.56	8.27	35.5	2.91	38.5	53.2	3.80	53.9	64.9

[a] third scenario from Leip et al., (2010).

[b] contribution % of purchased concentrate feed to global warming potential

[c] difference % between no-LUC scenario and LUC scenario

There is a small difference between GWP of “BIO” (no land use change) and the “Scenario 1”. “Scenario 2” had 24.0% (±9.37) higher emission compared to the baseline scenario, while the highest impact was achieved in “Scenario 3” with 2.58 (±0.61) CO_2-eq. kg^{-1} FPCM (36.1% (±13.1) higher emission compared to “BIO”). No difference in GHG emission among the four scenarios occurred in case of concentrate feed without soybean or soymeal (i.e. corn meal, corn and barley meal mix, etc.) were purchased. Flysjö et al. (2012) found a significant increment of GWP of milk production for both two groups of Swedish organic and conventional farms when different levels of LUC emissions (deforestation) due to soymeal imported from South America were considered, moreover in the Swedish study the conventional system used more soymeal per kg of milk resulting in a general higher CF compared to the organic system. The average contribution from concentrate feed to greenhouse gases emission of milk varied from 19.6% (±6.22) of “BIO” to 22.5% (±6.81) of “Scenario 1” and 38.6% (±10.5) of “Scenario 2” (table 3), but when LUC emission proposed by Leip et al., (2010) (Scenario 3) was addressed to Brazilian soy, the contribution of global warming potential from concentrate feed production raised value of 48.3% (±13.0) of total impact. The high variability in each scenario is due to the different amount of concentrate feed purchased by each farm and also to the composition (formula) of each feed considered. Mogensen et al., (2012) in a recent study showed that the GHG emissions related to feed production calculated per kg milk produced changed widely from a ‘local’ (all feeds grown in Denmark and the concentrated protein feed is based on rapeseed cake and cereals) to an ‘import’ (the concentrated protein feed is based on imported soybean meal and cereals) strategies when direct land use change emission from soymeal was considered: the impact from feed production was 0.22 CO_2-eq. kg^{-1} ECM for the ‘local’ strategy and 0.48 CO_2-eq. kg^{-1} ECM for the ‘import’ strategy.

In Figure 3 are graphically represented the results carried out from the PCA. The first dimension explains 36.7% of the total variation while the second dimension explains 19.0%. The two values of global warming potential are plotted opposite in the graph. The analysis showed that the variables enclosed in the oval figure on the right side (called “INTENSIVE FARMING”) had a positive effect in mitigating the GWP of “BIO” whereas an opposite trend was observed in relation of the share of summer grazing animals and the quota of pasture land on total farm land (oval figure on the left side called “EXTENSIVE FARMING”). High stocking rate seemed to be detrimental for the environmental point of view, moreover farms with high stocking rate were forced to buy more feed from the external market, that explained the opposite position between stocking rate and feed self-sufficiency. The feed self-sufficiency is also opposite on the second dimension to the GWP of "Scenario 3", when the impact was evaluated addressing high value of LUC to soybean production, the contribution of purchased concentrate feed became relevant. The closeness between "EXTENSIVE FARMING" and stocking rate would be contradictory but an explanation is that the stocking rate was estimated on the base of lowland area and there was a tendency of some farms (especially with high LU ha^{-1}) to compensate the lack of lowland for crop production with more pasture land. The analysis showed a strong positive correlation between maize silage % DMI, maize silage % lowland, feed efficiency, IOFC and milk production. When the impact of "Scenario 3" was considered the global assessment changed widely. The farm characteristics which had a mitigation effect in the baseline scenario lost their relevance, whereas the farming practices linked to a more extensive and low-input system ("EXTENSIVE FARMING") seemed to be more environmental sustainable reducing the GWP of milk production.

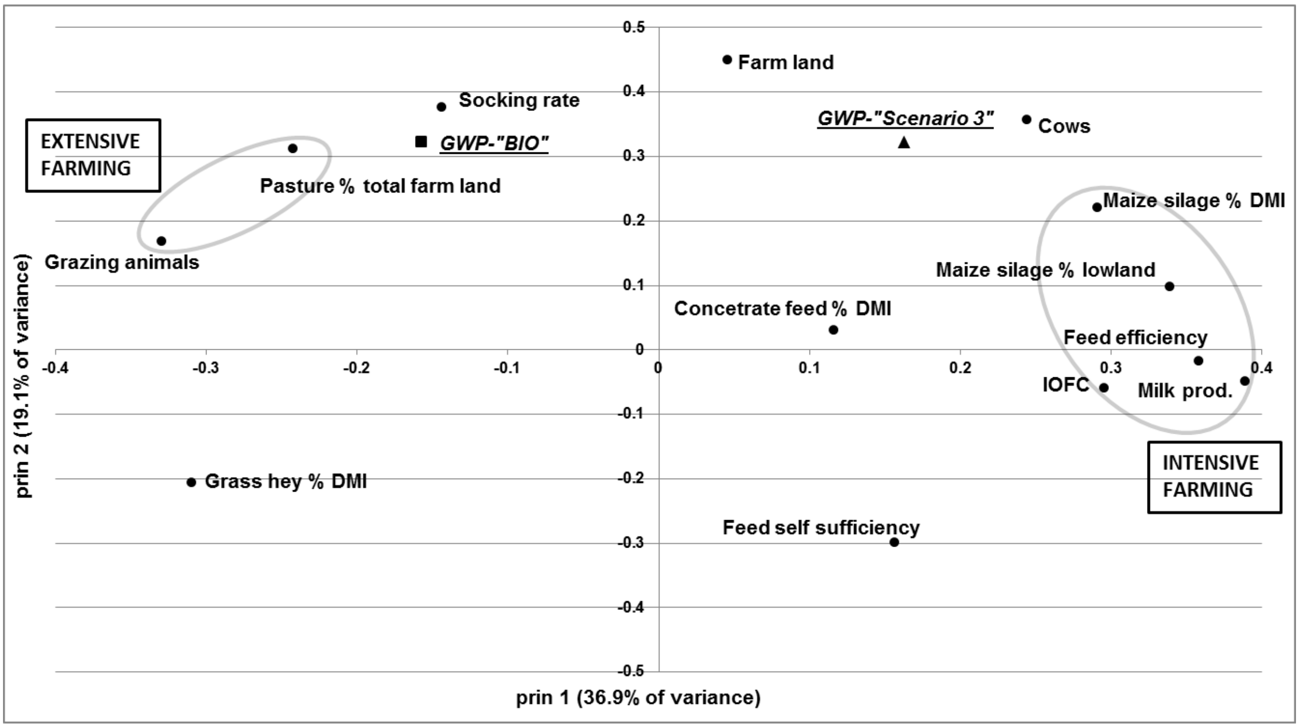

Fig. 3 - Principal Component Analysis (PCA).

The PCA confirmed the positive effect of some farm characteristics in improving the environmental performances, while for other variables the relation was not so clear.An high efficiency in feed conversion was recognized to play an important role in mitigation of greenhouse gas emission of livestock production (Hermansen and Kristensen, 2011; Opio et al., 2012), the positive effect of high production level on the environmental impact of 1 kg of milk was underlined by Capper et al. (2008) and Gerber et al. (2011). A different trend was shown by Vellinga et al. (2011) since great milk production was associate to an high feed ingestion and consequently high enteric emissions. Moreover, in a system expansion perspective, the conclusion that increasing milk yield in dairy production leads to a lower CF for milk is no longer obvious (Zehetmeier et al., 2011, Flysjö et al. 2012). Growing maize for silage instead grass could be a strategy to improving efficiency in crop production, but depending from cultural practice (high utilization of pesticides and fertilizer), generally grass need less input then maize. The stocking rate could heavily affect the environmental impact on the local area (farm nutrient surplus), the feed self-sufficiency is generally negative related with the amount of concentrate feed purchased which could significantly contribute to the GWP of milk production, especially if LUC of soymeal is considered. Potential feeding strategies to reduce CH_4 enteric emissions in ruminants are replacing grass by maize silage and increase the ratio of concentrates over roughage but the net GHG reduction along the chain is not self-evident (de Boer et al., 2011). The grazing summer period might be considered an environmental sustainable activity due to the few external inputs (i.e. none chemical fertilizers are used and only limited amount of commercial feed is allowed) but for some farms it was also a way to achieve more land in order to dilute the livestock density. It was clear that this strategy was not sustainable because the grazing period is limited to three month while for the rest of the year the herd is confined in the barns on lowland.

3.4. Milk production in mountain area and environmental impact

The results of this study showed an high average GWP of milk production in mountain area and also in a local perspective the environmental burden was relevant as shown by the farm nutrient balance. In general the structural and geographical features of the farms makes difficult to achieve a good environmental performance: the low milk yield associate to a low feed conversion rate were clearly hot spots. Moreover the value of feed self-sufficiency observed forced the farms to buy concentrate feed which can heavily weight on the total GHG emission and have a negative effect on the nutrient surplus. Some farms bought also forages from the market, probably because, over a poor land availability, the climatic condition of these area did not allow to have high yield per hectare and not always products of good nutritional quality which could also affect the feed efficiency. The activity which are traditionally related to extensive farming (i.e. grazing summer period in high pastureland) did not show any mitigation effect on the CF. Overall the huge variability observed in the results suggests that should be possible to achieve a more sustainable dairy production.

3.5. Sensitive analysis

The sensitive analysis for different allocation methods and for different LUC values underlined that one of the LCAs limits is that changing some basic assumption might change the overall results, moreover the reliability of the data should be taken into account when this kind of analysis are performed. That explains also why a comparison between LCAs studies, even on the same product, should be done carefully.

The ISO 14044 standards (ISO, 2006) suggests, wherever possible, to avoid allocation at least expanding the product system to include the additional functions related to co-products. In these study the system expansion was limited to the farm products (milk and meat) but in a global assessment should be more correct to use this procedure in all the processes included in the analysis. However this approach, known as Consequential Life Cycle Assessment (CLCA), is more complex to carry out especially in a multi process system like dairy farming. Anyway, to better evaluate the effects of a system expansion between milk and meat, a more precise value for Italian beef production systems (i.e. finishing beef bulls, dairy bull calf or suckler cow-calf) is needed.

The physical allocation method proposed by IDF (2010), compared to an economic allocation, should be preferred as proposed by the ISO standards (ISO 14044 (ISO, 2006)), moreover the IDF methodology is not influenced by the variability of the prices over the years.

This sensitive analysis on LUC soybean production clearly explained that including or not this emission significantly affected the final results (CF of milk). Moreover, the relevance of some system variables, in relation to the environmental impact, became contradictory.

3.6. Further improvements

Due to the high share of grassland on the total farm land the carbon sequestration should be included in this kind of analysis. The sequestration potential by grasslands and rangelands could be used to partly mitigate the greenhouse gas (GHG) emissions of the livestock sector, moreover a pasture-based systems might contribute to maintaining marginal grasslands and therefore contribute to soil C sequestration (Soussana et al., 2010). However the relation between livestock production and soil C sequestration is complex and uncertain (de Boer et al., 2011).
Over the past centuries the dairy farming in Alpine area had a basic role in preserving biodiversity richness not only for the grazing activity in high mountain pasture but also trough the management of semi-natural grassland of the medium altitude and lowland. It is universally recognized that mountain pastures host several species of plants and wild animals (Parolo et al., 2011) for that reason in the future will be necessary to include a global assessment of the impact on biodiversity richness of milk production in Italian alps.

4. CONCLUSIONS

This work showed that the common perception of which dairy farming systems in mountain area are "environmental friendly" does not reflect a real situation both in a global product perspective and in a local scale.
From the other point of view, still now, it is proven that this kind of agricultural activity in marginal areas plays an important role in preserve the typical landscape of these territories (Loszach et al., 2008) that every year is under an increasing risk of abandonment and degradation. Especially grazing and transhumance are of particular importance for the preservation of open landscapes as well as sustain rural communities in the European mountains (Leip et al., 2010) which base part of their earnings on the production of high quality typical products. Moreover natural and semi-natural alpine grassland are biodiversity hotspots. A good farm management is strategic to maintain the species richness of lowland grassland, but in particular the practice to transfer grazing animals in alpine pastures during summer period is recognized to positively influence the plant biodiversity (Parolo et al., 2011).

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WORK 4: ***HOW DIFFERENT LEVELS OF INTENSIFICATION CAN AFFECT THE ENVIRONMENTAL IMPACT OF MILK PRODUCTION? A STUDY ON ITALIAN FARMS.***

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ABSTRACT

Intensive dairy systems are associated with high stocking rates, high use of chemical fertilizers and pesticides, and mechanized methods, which often lead to problems of direct point source pollution, diffuse pollution and pressure on marginal habitats and landscape features (EC, 2000). In North Italy the livestock sector have e relevant impact on the use of natural resources due to the high animal concentration. The aim of this study was to assess the environmental impact of dairy production in a life cycle approach and to identify relations between different farming intensities and environmental performances. 29 dairy farms located in Po Valley were involved in the study, data collected during personal interview to the farmers were analyzed in order to estimate on- and off-emissions, over global warming potential, acidification potential, eutrophication potential, non-renewable energy use and land use were included as environmental impact indicators. The results carried out from the LCA were comparable with other recent studies and they showed an high variability among the sample of farms. A cluster analysis was performed in order to identify farming systems differing for the level of intensification, three clusters of farms were obtained. No statistically significant differences regarding the environmental performances on product base were observed among the groups, whereas, in a local perspective, higher nitrogen losses were associated to more intensive systems. This study pointed out that it is not clearly identifiable which is most environmental friendly way to produce milk, especially in a sample of farm belonging to the same system (conventional) despite some significant managing and structural differences among the units.

Key words: LCA, milk, intensive farming, environmental impact.

1. INTRODUCTION

During the last decades the European livestock sector showed a general trend in increasing intensification and enlarging the farm units, but diversity of farming systems remains mainly due to the biophysical conditions in different regions of Europe (Leip et al., 2010). The intensification of production is generally based on increasing of the stocking rate, on breeding genetically improved dairy cattle and on increasing in concentrates in the diet (Alvarez et al., 2008). Also in Italy dairy farming system is showing a progressive intensification: although the number dairy cattle decreased in the last 30 years (from 2.6 million in 1980 to 1.6 million) the number of dairy cow for farm increased from 7.9 to 31.8 in the same years (ISTAT, 2012). Moreover in Northern Italy favorable climate and infrastructure conditions led to a significant livestock concentration (84% of total Italian dairy cows with an average number of dairy cows per farm of 75 (AIA, 2011)) with a consequence intensive utilization of natural resources (i.e. land, air, water) and high environmental pressure. Intensification of milk production system could have a negative effect on the environment: a study (Basset-Mens et al., 2009) conducted in New Zealand showed that increasing the number of cows per land unit (with an higher N-fertilization and more land used to grow corn for silage) was detrimental to dairy farms' eco-efficiency in terms both of milk production and land use functions. Also Penati et al. (2011) found that the best environmental performance was obtained by a group of alpine farms characterized by: low stocking density and production intensity, high feed self-sufficiency and lowland availability. Casey and Holden (2005) suggested that, to improve the environmental efficiency of dairy farms, a move toward fewer cows producing more milk at lower stocking rates is required, such a move would represent extensification in terms of area but intensification in terms of animal husbandry. On the other hand a reviewed study of Crosson et al. (2011) concluded that increased output/ha through increased intensification can reduce emissions/kg product, provided that excessively high levels of N fertilizer use can be avoided and that overall emissions associated with intensification are offset by higher levels of productivity. Kristensen et al. (2011) identified the "herd efficiency" and the "farming intensity" to be relevant strategies for environmental impact reduction. However it is still debated and under study which are the best farming systems and technical strategies for the potential mitigation of environmental impact of dairy farms.

LCA is a method globally accepted to calculate the environmental impact of agricultural products. The predominant environmental consequences quantified in LCA studies of dairy systems are the acidifying and eutrophic effects of milk production on watercourses, the global warming effect of dairy systems and the utilization of resources such as land and non-renewable energy during the production of milk (O'Brien et al., 2012).

Over the estimation of the environmental impact of milk production at the farm level through a LCA approach, the objective of this study was: i) to statistically analyze the relation between impact values and farm characteristics ii) on the base of these relationship identify dairy farming systems with different levels of production intensity iii) and to highlight relevant differences in environmental performances among the system.

2. MATERIALS AND METHODS

2.1. System descriptions

29 dairy farms were involved in this study. All these farms were located in Northern Italy and they were members of a cheese factory which produced Grana Padano cheese O.P.D.

Farm activities like growing of forages and other crops, manure storages and management, livestock housing and energy consumption (electricity and diesel) plus off-farm activities linked to production of external inputs like fertilizer, pesticides, fodders and raw materials purchased, concentrate feed, breeding of replacing animals and production of litter materials (straw and sawdust) were considered in the analysis. Production of some inputs, for instance medicine, were excluded because of their small effect on the environmental impact of milk production (O'Brien et al, 2012). All data were referred to year 2010.

The functional unit (FU), which describes the primary function fulfilled by a product system, was established as 1 kg fat protein corrected milk (FPCM) leaving the farm gate (Thomassen et al, 2008) and estimated using the formula: FPCM (kg) = (0.337 + 0.116 x % fat + 0.060 x % protein), from Gerber et al. (2010). As the dairy farm is a multifunctional system an allocation between the different outputs is required. In this study a biological allocation developed by IDF (2010), was used. This allocation method is based on the feed energy required to produce the amount of milk and meat at the farm: AF = 1 - 5.7717 x R where AF = allocation factor for milk, R = M_{meat} / M_{milk}, M_{meat} = sum of live weight of all animals sold included bull calves and culled mature animals and M_{milk} = sum of milk sold. Allocation factor for meat is: 1 - AF (allocation factor) to milk.

2.2. Data collection

All the data considered to be relevant for the impact estimation were collected through a personal interview to the farmers. The questions were addressed to get precise information about the crop systems and field operation, fuel consumption, number of animals and the housing systems of the different animals categories, storages management and animal feeding strategies. Over these information, other data about the amount of purchased feeds (both roughages and concentrates), purchased fertilizers and pesticides, purchased bedding materials and the number and the origin of purchased replacing animals were collected during the interview.
Moreover, during the farms visits, forages (hays and silages) and the Total Mix Ration (TMR) were sampled. Forages and TMR were analyzed for content of DM, ash, CP, ether extract and crude fiber (CF) with methods of AOAC (1995) and starch with method AOAC (1998); NDF was analyzed with method of Mertens (2002), ADF and ADL with method of Van Soest et al. (1991). The data obtained from these analysis were used in the estimation of pollutant emitted at the farm level. The amount of milk produced by each farm was provided by the cheese factory whereas the amount of meat (as animal live weight) was estimated on the base of the number of animal sold for slaughter and their live weight declared by the farmers.
The composition of each concentrate feed was estimated on the base of the raw materials reported on the commercial label with the help of CPM-Dairy Ratio Analyzer Beta V3 software (Cornell-Penn-Miner, 2004).
In Table 1 is summarized an inventory of the most important data used for impact assessment. The data are expressed as the average value of the 29 dairy farms.

Tab. 1 - Inventory data (average 29 farms).

	Unit	Mean	Standard Dev.	Minimum	Maximum
LAND					
Farm land	ha	40.0	27.6	8.50	120.0
Permanent grassland	% of farm land	50.3	25.3	0.00	100.0
Maize for silage	% of total land	38.7	23.1	0.00	100.0
EnL land productivity	MJ ha^{-1} farm land	75842	34901	134248	21082
N land productivity*	kg ha^{-1} farm land	178.5	103.4	237.6	38.0
N synthetic fertilizers	kg ha^{-1} farm land	90.2	63.4	0.00	282.4
Pesticides (a.s.)**	g ha^{-1} farm land	838	0	2140	653
HERD					
Dairy cows	n	93.2	53.5	17.00	195.00
Livestock Unit	n	149	92.3	25.7	335.00
Milk production	kg $farm^{-1}$ $year^{-1}$	794756	505455	135763	1867522
Milk protein	%	3.43	0.15	3.02	3.75
Milk fat	%	3.94	0.15	3.68	4.28
Beef production (as live weight sold)	kg $farm^{-1}$ $year^{-1}$	19721	2550	70250	16157
STORAGES					
Solid manure	%	40.72	0.00	100.00	37.40
Liquid slurry	%	59.28	0.00	100.00	37.40
FEED					
Feed produced on-farm	t DM $year^{-1}$	532.9	81.7	1733.0	387.5
Purchased forages	t DM $year^{-1}$	73.3	0.0	519.76	119.0
Purchased concentrates	t DM $year^{-1}$	250.5	25.0	723.0	181.3
ENERGY					
Diesel use	kg LU^{-1} $year^{-1}$	87.9	54.0	141.1	21.3
Electricity use	kwh LU^{-1} $year^{-1}$	208.0	52.3	336.2	80.1

* crops nitrogen yield
** active substance

Other indicators like stocking rate, replacing rate, feed self-sufficiency, feed conversion rate were estimated on the base of data collected. Moreover the farm nutrient balance was calculated because it provides an estimation of the nitrogen burden on the farm land and it was estimated in accordance of Penati et al., (2011), while the income over feed cost (IOFC) was used as economic indicator of farm performances as proposed by Hutjens, 2007.

2.3. Emissions estimation

2.3.1. Greenhouse gases (GHG) emissions at farm level

The substances which mainly contributes to the global warming potential of livestock production are CO_2, CH_4, N_2O (Steinfeld et al., 2006).

Methane (CH_4) emissions from livestock enteric fermentations were estimated using the equation (equation [8d] R^2=0.63) from Ellis et. al. (2007) considering the amount of dry matter intake (DMI) and the content of NDF and ADF of the diet. The factor 55.65 (MJ/kg CH_4) (IPCC 2006a) was used to convert the energy of enteric methane in kilograms of methane emitted.
Methane emissions from manure management were estimated using Tier 2 method suggested from IPCC (2006a). Volatile solid excretion (VS) is the parameter which mainly influences the methane emissions from manure management and it was estimated considering:

- Gross Energy (GE) of the diets (kj kg^{-1} DM) evaluated using the equation from Ewan (1989) which required the content of CP, EE and Ash of the diet obtained from the laboratories analysis of the ration;
- Digestibility of the feed (DE%) estimated using a calculation model developed for each type of forages and concentrate feed on the base of the equation proposed by INRA (2007). The feed nutritional characteristics were obtained from the laboratories analysis.

The methane conversion factors for each manure management system were 4% for solid storage, 17% for liquid slurry and 27% pit storage.
In this study animal N excretion was estimated as proposed by the IPCC (2006a) Tier 2 method considering the amount of nitrogen intake (on the base of % CP of diet) subtracting the nitrogen retained by the animals. The nitrogen retained was evaluated as the nitrogen output with milk production plus the nitrogen stored in the animal bodies during the growing period. For the estimation of nitrogen stored in the animal bodies was considered the Net Energy for Growth. It was calculated using the NRC (1996) equation and default values of body weight (kg), mature body (kg) and weight gain (kg) which were taken from the CPM-Dairy Ratio Analyzer Beta V3 software (Cornell-Penn-Miner, 2004) on the base of animal age.
Nitrous oxide (N_2O) emissions from manure storages occurred in direct and indirect form and in both cases they were estimated using the Tier 2 method from IPCC (2006a). Emission factors used to quantify direct N_2O-N losses from manure storages were: 0.005 (0.0027-0.01) for solid storage, 0.005 for liquid slurry and 0.002 for pit storage.
Indirect N_2O emissions related to N volatilized during manure storage were estimated using the emission factor: 0.01 (0.002-0.05) kg N_2O-N kg^{-1} N volatilized. Total N volatilized was quantified as 0.3 (0.1-0.4) for solid storage, 0.4 (0.15-0.45) for liquid slurry and 0.28 (0.1-0.4) for pit storage.
Direct and indirect N_2O losses from fertilizers application were estimated following the Tier 1 method suggested from IPCC (2006b): over the amount of nitrogen applied to the soils from synthetic fertilizers and manure (slurry and solid) the nitrogen from crop residues was accounted in the estimation. The emission factor used for direct N_2O emissions was 0.01 (0.003-0.03)) kg N_2O-N kg^{-1} N applied. Indirect N_2O emission at field level occurred after nitrogen volatilization and leaching. Indirect N_2O losses were 0.01 (0.002-0.05) kg N_2O-N kg^{-1} N volatilized. N volatilized was quantify to be 0.1 (0.03-0.3) and 0.2 (0.05-0.5) of nitrogen applied in form of synthetic and organic fertilizer respectively. Other indirect N_2O emissions were estimated to be 0.0075 (0.0005-0.025) kg N_2O-N kg^{-1} N leached. Nitrogen leached in the soil was supposed to be 0.3 (0.1-0.8) of total nitrogen applied. The conversion factor (molecular weights) from N-N_2O to N_2O is 44/28.
CO_2 emissions from fuel combustion were estimated on the base of fuel consumption for each farm. Emissions occurred during field operations (i.e. plowing, harrowing, sowing, harvesting, etc.) were estimated on the base of data collected and modeled in the Simapro PhD 7.3.3 software (Prè Consultants, 2012) using the processes of the Econivent (2007) database. To estimate CO_2 emissions related to other fuel consumptions (i.e. the use of feeding mixer) the emission factor used was 3.12 kg of CO_2 kg^{-1} of diesel proposed by Nemecek and Kägi (2007). Models and emission factors used for on-farm GHG emission estimation are detailed in Table 2. Emissions from livestock respiration and the variation in soil carbon stock were not accounted.

Tab. 2 - Models and emission factors used for the estimation of GHG emissions at the farm level.

Pollutant	Source	Amount	Reference
CH_4	enteric	CH_4 (MJ) = 2.16 (± 1.62) + 0.493 (± 0.192) * DMI (kg) − 1.36 (± 0.631) * ADF (kg) + 1.97 (± 0.561) * NDF (kg)	Ellis et al. (2007)
	storages	CH_4 = VS * B_0 * 0.67 * MCF/100 * MS	Eq. 10.23 - IPCC (2006a)
		VS = [GE * (1-DE/100)+(UE * GE)] * [(1-Ash)/18.45]	Eq. 10.24 - IPCC (2006a)
		GE (kj) = 17350 + (234.46 * EE%) + (62.8 * CP%) - (184.22 x Ash%)	Ewan (1989)
		DE: feed digestibility	Tab. 8.6 - INRA (2007)
		MCF solid storage: 4	IPCC (2006a)
		MCF liquid slurry: 17	
		MCF pit storage: 27	
N_2O direct	storages	N_2O = Nex * MS * EF * 44/28	Eq. 10.25 - IPCC (2006a)
		Nex = Nintake * (1− Nretention)	Eq. 10.31 - IPCC (2006a)
		N intake: DMI * (CP%/100/6.25)	
		N retention: N retained per animal with milk and weight gain	Eq. 10.33 - IPCC (2006a)
		EF solid storage: 0.005 (0.0027 - 0.01)	Tab. 10.21 - IPCC (2006a)
		EF liquid slurry: 0.005	
		EF pit storage: 0.002	
	field	N_2O = (Nsn + Non + Ncr) * EF * 44/28	Eq. 11.2 - IPCC (2006b)
		Non: annual amount of N from managed animal manure applied to soil (Nex - Frac_loss + N bedding)	Eq. 10.34 - IPCC (2006a)
		Frac_loss solid storage: 40% (10 – 65)	Tab. 10.23 - IPCC (2006a)
		Frac_loss liquid slurry: 40% (15 – 45)	
		Frac_loss pit storage: 28% (10 – 40)	
		EF: 0.01 (0.003 - 0.03)	Tab. 11.1 - IPCC (2006b)

Tab. 2 - Follows.

N_2O indirect	storages	$N_2O_{(G)}$= Nvolatilization * EF * 44/28	Eq. 10.27 - IPCC (2006a)
		Nvolatilization: Nex * MS * Frac_GasMS/100	
		Frac_GasMS solid storage: 30 (10 – 40)	Tab. 10.22 - IPCC (2006a)
		Frac_GasMS liquid slurry: 40 (15 – 45)	
		Frac_GasMS pit storage: 28 (10 – 40)	
		EF: 0.01 (0.002 - 0.05)	Tab. 11.3 - IPCC (2006b)
	field	$N_2O_{(ATDN)}$ = [(Nsn * Frac_GasF) + (Non * Frac_GasM)] * EF * 44/28	Eq. 11.9 - IPCC (2006b)
		Frac_GasF: 0.1 (0.03 - 0.3)	Tab. 11.3 - IPCC (2006b)
		Frac_GasM: 0.2 (0.05 - 0.5)	Tab. 11.3 - IPCC (2006b)
		EF: 0.01 (0.002 - 0.05)	Tab. 11.3 - IPCC (2006b)
		$N_2O_{(L)}$ = (Nsn + Non) * Frac_Leach * EF * 44/28	Eq. 11.10 - IPCC (2006b)
		Frac_Leach: 0.3 (0.1 - 0.8)	
		EF: 0.0075 (0.0005 - 0.025)	Tab. 11.3 - IPCC (2006b)
CO_2	field operations		Econivent (2007)
	diesel combustion[a]	CO_2 = kg diesel * EF	
		EF: 3.12 kg of CO_2 kg^{-1} of diesel	Nemecek and Kägi (2007)

[a] excluding the quota used during field operations

2.3.2. Other emissions at farm level

Ammonia (NH_3) and nitrogen oxides emissions (NOx) that occur during animal housing, manure storages and spreading were estimated following the method proposed by EAA (2009a,b) on the base of the total amount of nitrogen excreted by the animals. The Tier 2 used a mass flow approach based on the concept of a flow of TAN (Total Ammoniacal Nitrogen) through the manure management systems. The proportion of TAN was estimated to be 0.6 of nitrogen excreted by the animals. NH_3-N emission factors, as proportion of TAN, were specific for each step in manure handling and manure types (slurry or solid) (EAA 2009a). NH_3-N emission factors for housing were 0.2 and 0.19 for slurry and solid respectively, for storages were 0.2 and 0.27 for slurry and solid respectively and for spreading were 0.55 and 0.79 for slurry and solid respectively (EAA 2009a). The total amount of NH_3 volatilized was obtained multiplying NH_3-N by the conversion factor 17/14 (ratio between molecular weights of NH_3 and N).

The fraction of NO-N lost during manure storages was estimated to be 0.0001 and 0.01 of TAN for slurry and solid respectively (EAA 2009a) and then converted in NOx (NOx/N = 30/14) whereas NOx emitted during manure spreading was estimated using the default emission factor 0.026 kg NOx kg^{-1} fertilizer-N applied as proposed by Tier 1 of EAA (2009b). NH_3 and NOx emissions related to crop fertilization with synthetic fertilizers were estimated following EAA (2009b) guidelines. In the estimation of NH_3 volatilization, the Tier 2 was adopted (EAA 2009b). In this approach the NH_3 emission factors were specific for each type of fertilizers and it also accounted for the average spring temperature (°C): NH_3 emission factors used (expressed as kg NH_3 kg^{-1} synthetic N applied

to the soil) were 0.1067 + 0.0035 * t_s; 0.0080 + 0.0001 * t_s and 0.0080 + 0.0001 * t_s for urea; ammonium nitrate and N:P:K fertilizers respectively and t_s was the mean spring temperature. NOx lost after the application of synthetic fertilizers were estimated following the Tier 1 methodology of EAA (2009b) using the default emission factor 0.026 kg NOx kg^{-1} fertilizer-N applied.
The amount of nitrogen leached was estimated using the emission factor 0.3 (0.1-0.8) kg N kg^{-1} N applied to the field (IPCC 2006b) and then converted in NO_3 using the ratio of molecular weights (NO_3/N = 62/14). To estimate emissions of PO_4^{3-} were considered the amount of phosphorus loss in dissolved form to surface water (run-off) and leached as proposed by Nemecek and Kägi, (2007), and converted in phosphate with the coefficient 95/31. This method considers the amount of phosphorus excreted by the animals and applied to the field and also the input from chemical fertilizers. The method took into account the quota of land that is arable or meadow. Table 3 reports the models used for the estimation of acidifying and eutrophic substances emitted at farm level.

Tab. 3 - Models and emission factors for the estimation of ammonia, nitric oxide and phosphate emissions

Pollutant	Source	Amount	Reference
NH_3	housing	TAN = Nex * EF_TAN	Eq. 10 - EEA (2009a)
		EF_TAN: 0.6	Tab. 3-8 - EEA (2009a)
		NH_3build_slurry = TANbuild_slurry * EFbuild_slurry * 17/14	Eq. 15 - EEA (2009a)
		EFbuild_slurry: 0.2	Tab. 3-8 - EEA (2009a)
		NH_3build_solid = TANbuild_solid * EFbuild_solid * 17/14	Eq. 16 - EEA (2009a)
		EFbuild_solid: 0.19	Tab. 3-8 - EEA (2009a)
	storages	NH_3storage_solid = TANstorage_slurry * EFstorage_slurry * 17/14	Eq. 29 - EEA (2009a)
		EFstorage_slurry: 0.20	Tab. 3-8 - EEA (2009a)
		NH_3storage_solid = TANstorage_solid * EFstorage_solid * 17/14	Eq. 30 - EEA (2009a)
		EFstorage_solid: 0.27	Tab. 3-8 - EEA (2009a)
	field	NH_3applic_slurry = TANslurry_applic * EFapplic_slurry * 17/14	Eq. 35 - EEA (2009a)
		EFapplic_slurry: 0.55	Tab. 3-8 - EEA (2009a)
		NH_3applic_solid = TANsolid_applic * EFapplic_solid * 17/14	Eq. 36 - EEA (2009a)
		EFapplic_solid: 0.79	Tab. 3-8 - EEA (2009a)
		NH_3applic_fert =Nfert_applic * EFfert_type	Eq. 3 - EEA (2009b)
		EFurea: 0.1067 + 0.0035 * Ts	Tab. 3-2 - EEA (2009b)
		EFamm.nitr. and NPK: 0.0080 + 0.0001 * Ts	
NOx	storages	NOx storage_solid = TANstorage_slurry * EFstorage_slurry *17/14	Eq. 29 - EEA (2009a)
		EFstorage_slurry: 0.0001	Tab. 3-9 - EEA (2009a)
		NOxstorage_solid = TANstorage_solid * EFstorage_solid * 17/14	Eq. 30 - EEA (2009a)
		EFstorage_solid: 0.01	Tab. 3-9 - EEA (2009a)
	field	NOxapplic_tot= (Nslurry_applic + Nsolid_applic + Nfert_applic) * EFapplic	
		EFapplic: 0.026	Tab. 3-1 - EEA (2009b)
PO_4^{3-}	field	P_{gw} (leached to ground water) = P_{gwl} * F_{gw}	Par. 4.4.3 - Nemecek et al. (2007)
		P_{gwl} arable land: 0.07	
		P_{gwl} permanent pasture and meadow: 0.06	
		F_{gw}: 1+0.2/80 * P_2O_5slurry	
		P_{ro} (P lost through run-off to rivers) = P_{rol} * F_{ro}	
		P_{rol} open arable land: 0.175	
		P_{rol} ixtensive meadow: 0.25	
		Frofert: 0.2/80 * P_2O_5fert	
		Froslurry: 0.7/80 * P_2O_5slurry	
		Fromanure: 0.4/80 * P_2O_5manure	

2.3.3. Off-farm processes

The emissions related to off-farm activities were calculated using LCA software, Simapro PhD 7.3.3 (Prè Consultants, 2012) and were modeled mainly using the Ecoinvent (2007) database. The following processes were considered: the production chain of commercial feed (from crop growing to feed factory processing), production of forages purchased, production of bedding material,

process to rear the replacing animals purchased, the production of chemical fertilizers and pesticides, the production of diesel and electricity used in the farms.
The transportation were accounted only for the feed, bedding materials and purchased replacing animals. In Table 4 are reported the off-farm processes considered in this study and the relative references.

Tab. 4 - Off-farm inventory data.

Process	References
FEED PRODUCTION	
Crops	Ecoinvent, 2007; Baldoni e Giardini, 2002; Ribaudo, 2002; data from this study
Milk powder	LCA food DK, 2007
Feed processing	LCA food DK, 2007
FORAGES PRODUCTION	Ecoinvent, 2007; Baldoni e Giardini, 2002; Ribaudo, 2002; data from this study
BEDDING MATERIAL PRODUCTION	Ecoinvent, 2007
REARING ANIMALS	data from this study
FERTILIZERS PRODUCTION	Patyk and Reinhardt, 1997; Ecoinvent, 2007;
PESTICIDES PRODUCTION	Ecoinvent, 2007
ENEREGY PRODUCTION	Ecoinvent, 2007
TRANSPORTATION	Ecoinvent, 2007

2.4. Impact assessment

The environmental impact of each dairy farm was evaluated performing a detailed ‘‘cradle-to-farm-gate’’ partial LCA (Belflower et al., 2012). The environmental impact categories chosen were global warming, acidification, eutrophication, non-renewable energy use and land use (O'Brien et al, 2012) and their load was evaluated with the help of the LCA software Simapro. In particular the first four categories were estimated using the EPD 1.03 (2008) method, updated with IPCC 2007 GWP conversion factors (100 year time horizon) and set the value of CO_2 emission from land transformation to 0. Land use was evaluated using Ecological footprint (2009) method. Both methods are present in the software database. Despite a sensitive approach is recommended in a LCA study as some basic assumption might strongly affect the final outcomes (Flysjö, et al., 2011a), in this work no sensitive analysis was performed because it was not in the intent of the study.

2.5. Statistical analysis

Statistical analysis was performed using SAS 9.2 software (SAS, 2001). Principal Component Analysis (PCA, proc PRINCOMP in SAS (2001)) was used in order to study the relationships among several quantitative variables: total environmental impacts and the relative on- and off-farm contribution, production level (kg FPCM cow^{-1} day^{-1}), dairy efficiency (kg FPCM kg^{-1} dry matter intake), number of dairy cows, stocking rate (LU ha^{-1}), total farm land (ha) and share of maize silage and grassland on farm land. Moreover database was analyzed using the CLUSTER procedure (SAS, 2001) in order to identify different levels of farming intensity the following variables were considered: total farm land (ha), number of dairy cows, stocking rate (LU ha^{-1}), production level (kg FPCM cow^{-1} day^{-1}), quota of grass hey and maize silage on dry matter intake, share of maize silage on farm land, feed self-sufficiency, dairy efficiency (kg FPCM kg^{-1} dry matter intake). In the cluster analysis only 28 dairy farms were processed because one farm, considered to be an out layer, was excluded from the model. A GLM (SAS, 2001) analysis was performed to evaluate the differences between clusters identified.

3. RESULTS

3.1. Environmental impact

In Table 5 are reported the environmental impacts of the sample of the 29 farms involved in this study. The strong variation between results underline that beside high environmental sustainable farms there were other units which hardly achieved a good environmental performances.

Tab. 5 - Total environmental impacts expressed per kg of fat and protein corrected milk (FPCM) for the 29 dairy farms and on- and off-farm contributions.

Environmentl impact	Location	Mean	Standard dev.	Minimum	Maximum
Global warming (GWP), kg CO_2-eq.	Total	1.27	0.18	0.90	1.66
	On-farm %	*74.1*	*7.15*	*61.1*	*87.8*
	Off-farm %	*25.9*		*12.2*	*38.9*
Acidification, g SO_2-eq.	Total	15.29	3.34	8.63	21.71
	On-farm %	*86.6*	*5.48*	*70.1*	*94.9*
	Off-farm %	*13.4*		*5.11*	*29.9*
Eutrophication, g PO_4-eq.	Total	7.48	1.58	5.00	10.34
	On-farm %	*74.9*	*8.00*	*59.5*	*90.3*
	Off-farm %	*25.1*		*9.74*	*40.5*
Energy use, MJ	Total	5.47	0.94	2.85	7.33
	On-farm %	*42.9*	*12.4*	*22.9*	*72.1*
	Off-farm %	*57.1*		*27.9*	*77.1*
Land use, m^2	Total	1.55	0.35	0.74	2.06
	On-farm %	*47.6*	*14.3*	*27.4*	*82.3*
	Off-farm %	*52.4*		*17.7*	*72.6*

The average GWP was 1.27 ± 0.18 kg CO_2-eq. kg^{-1} FPCM. The on-farm share of greenhouse gas emissions was much higher compared to that off-farm. The most relevant contribution to GWP was from enteric and storages emissions (52.5% ± 4.26) followed by emissions related to the production of concentrate feed (19.8% ± 6.63). Almost all the impact to acidification was due to on-farm activities and the main role was played by farm crop production (39.4% ± 8.65), animal housing (22.6% ± 2.78) and manure storages (22.4% ± 5.26). The on-farm contribution to eutrophication was absolutely relevant in particular growing of farm crops was the major driver (52.2% ± 8.17) while in the off-farm processes the production of concentrate feed accounted for 20.8% ± 7.29 of total eutrophication potential. In the non-renewable energy use the on- and off-farm contributions were substantially equal, that is due to the production of concentrate feed which alone covered the 46.3% ± 13.6 of the total energy consume. Similarly to energy use the land use did not show a significant difference between the on- and off-farm shares, almost all the land use was related to crop production for animal feeding, totally it covered the 89.0% ± 4.89 of the global impact.
In Figure 1 are showed the contributions of different substances to GWP, acidification and eutrophication.

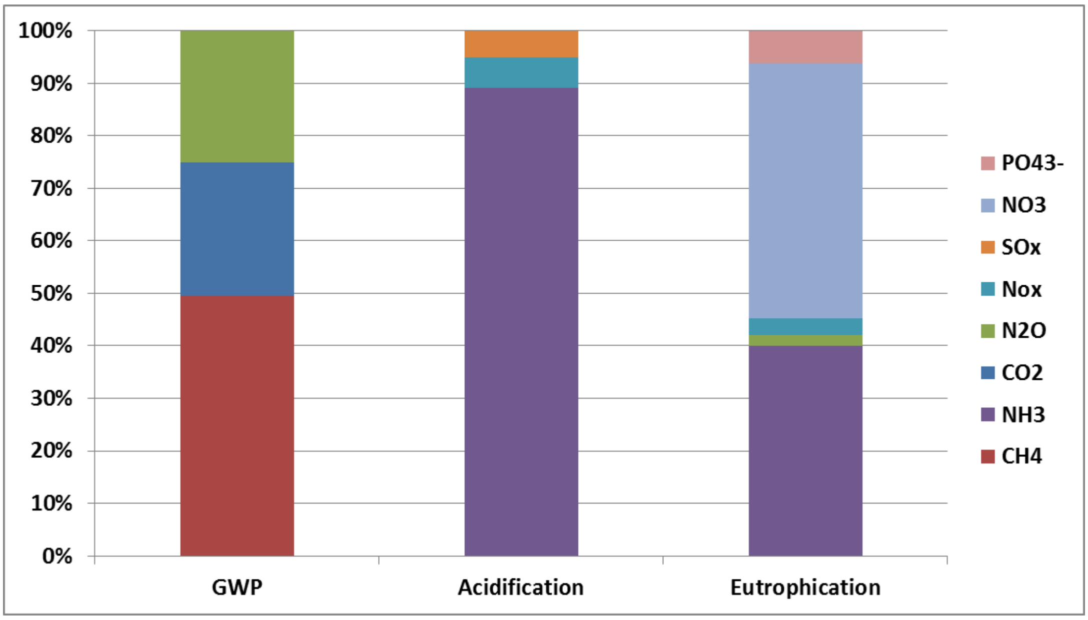

Fig. 1 – Contribution of different substances to the impact categories.

Overall methane was responsible of the 49.5% ± 3.59 of total greenhouse gases emission, followed by carbon dioxide and nitrous oxide which had the same weight (25.3% ± 2.70 and 25.1% ± 3.59 respectively). Globally enteric fermentation is the most relevant source of CH_4, in this study was found that 74.6% ± 8.81 of total methane was produced in the gastrointestinal tract of the animals.

Ammonia emission accounted for the 88.9% ± 3.59 of acidification potential, ammonia volatilized mainly during application of manure on farm soils (41.7% ± 9.47 of total ammonia emission) followed by losses occurred during animal housing and manure storages (25.4% ± 3.04 and 25.0% ± 5.67 of total ammonia emission). The quota of nitrate leached during crop production on-farm was higher than the off-farm fraction and they accounted for the 67.8% ± 10.4 and 27.5% ± 10.2 of total nitrogen leached respectively. The higher contributions to eutrophication potential were from NO_3 leaching (48.4% ± 4.08) and volatilization of NH_3 (39.9% ± 4.41) while the role of phosphates losses was less relevant (only the 5.97% ± 1.50).

3.2. Interaction between farm characteristics and environmental impact

In Figure 2 are plotted the results carried out by the principal components analysis. The first dimension explains 34.4% of the total variance while the second dimension explains 20.1%. All the global impact categories were high correlated each other, in fact they were pointed in the same dial (upper-right). The farm characteristics, which were enclosed in the upper-left dial, were inversely related with the global impacts. The distance on the first dimension between these two groups of variables suggested that an improvement of the farm features like milk production, dairy efficiency, stocking density and the share of grassland on farm land might result in a mitigation of all the impacts. Farms with high stocking density generally needed to buy more feed, for that reason stocking density and feed self-sufficiency were opposite in the graph. Moreover on-farm land use and energy use were related to feed self-sufficiency because higher was the quota of feed produced at the farm level higher was their contribution to total impacts. The two groups of "on-farm impacts" and "total impact" showed between them a relative short distance on the second dimension, moreover the second axis is less important in variance explanation. That means a positive correlation between the two groups of variable, especially for on-farm GWP, acidification and eutrophication which heavily contributed to the global environmental impact.

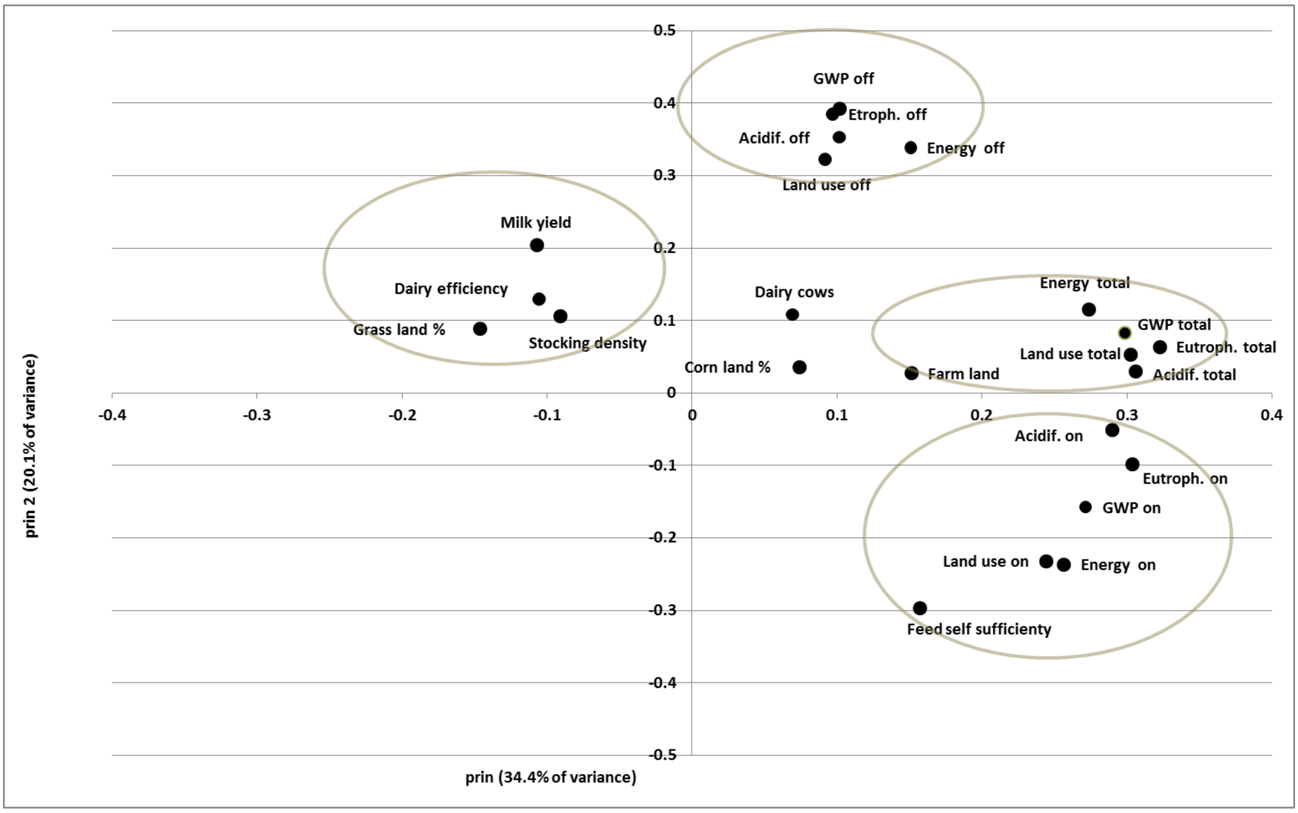

Fig. 2 - Principal component analysis for variables on the 29 farms (total, on-farm and off-farm environmental impacts, dairy cows, stocking density, milk yield, dairy efficiency, feed self-sufficiency).

3.3. Farming systems and environmental impact

The cluster analysis identified three different groups (Figure 3) , the first one ("Cluster 1") included 10 farms, in the second ("Cluster 2") were grouped 11 farms and the remaining 7 farms formed the third group ("Cluster 3").

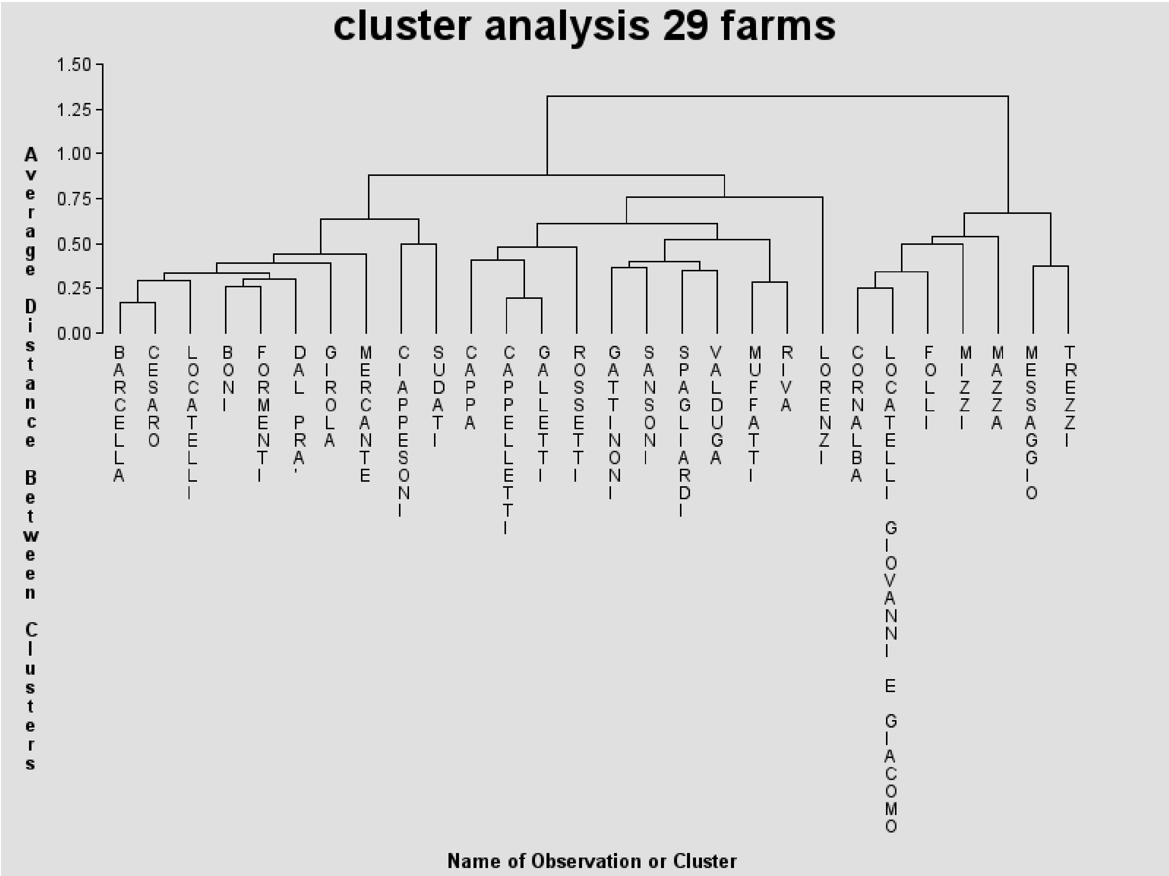

Fig. 3 - The tree procedure average linkage cluster analysis.

In Table 6 are reported the results obtained from the GLM analysis which was performed in order to identify how the farm characteristic varied among the three different groups.

Tab. 6 - Characteristics of clusters (Least square means - *p sistem < 0.1*).

Variable		Cluster 1	Cluster 2	Cluster 3	SE	P	P 1 vs. 2	P 1 vs. 3	P 2 vs. 3
Farms	n	10	11	7					
Farm land	ha	34.9	20.0	81.4	4.91	**<0.001**	**0.01**	**<0.001**	**<0.001**
Arable land	% of total land	54.1	34.5	61.7	8.18	**0.03**	**0.05**	0.48	**0.02**
Maize for silage	% of total land	45.7	26.9	38.5	7.26	**0.10**	**0.03**	0.45	0.22
Dairy cows	n	97.6	38.6	161	6.79	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Livestock Unit	n	157	56	258	11.9	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Stocking density	LU ha^{-1}	4.71	2.97	3.31	0.39	**<0.001**	**<0.001**	**0.01**	0.50
Replacing rate	%	26.1	25.9	26.3	4.26	1.00	0.96	0.97	0.93
Milk production	kg FPCM cow^{-1} $year^{-1}$	8827	7528	8576	455	**0.05**	**0.02**	0.68	**0.08**
Dry matter intake	kg cow^{-1} d^{-1}	21.2	20.0	21.2	0.67	0.21	0.12	0.99	0.16
Dairy efficiency	kg milk kg^{-1} DMI cow	1.36	1.22	1.33	0.06	**0.10**	**0.04**	0.66	0.15
Forage/concentrate		1.33	2.21	1.37	0.40	0.12	0.07	0.94	0.11
Maize silage intake	% DMI	30.4	22.8	34.7	4.46	0.11	0.15	0.47	**0.05**
Grass hay intake	% DMI	15.6	27.5	17.9	4.64	**0.09**	**0.04**	0.71	0.12
Feed self sufficiency	% total feed	54.6	71.7	69.2	4.76	**0.01**	**0.00**	**0.03**	0.68
N farm balance	kg/ha	757	389	600	84.6	**<0.001**	**<0.001**	0.17	0.06
N farm efficiency	%	18.0	23.8	16.0	2.39	**0.03**	**0.05**	0.53	**0.02**
Gross margin	€ cow^{-1} d^{-1}	4.81	4.41	5.01	0.62	0.730	0.583	0.804	0.456
IOFC	€ cow^{-1} d^{-1}	5.96	5.78	6.76	0.44	0.219	0.718	0.177	0.093

The quota of farm land showed a significant difference among all the groups with an higher value for “Cluster 3” and lower value for “Cluster 2”. The quota of arable land of “Cluster 2” was statistically lower compared to the other two groups moreover “Cluster 1” had the higher quota of land used to grow maize for silage. The number of dairy cows and total livestock units showed the same trend observed for the farm land and it suggests that overall the number of animals bred was proportionate with the land area. However the stocking rate is generally high especially in “Cluster 1” which was statistically different compared to the other groups. The production levels of “Cluster 1” and “Cluster 3” were significant higher compared to cluster “Cluster 2”, in the same way the conversion rate, which is strongly related to productivity, showed a better performance in “Cluster 1 and 3”. “Cluster 1” had the statistically lower feed self-sufficiency compared to the other groups which had similar values to each other. “Cluster 2” had significant lower farm nitrogen surplus and higher nitrogen farm efficiency, “Cluster 3”, despite the relative low stocking density, showed high value of nitrogen surplus that was because the lower nitrogen farm efficiency. Moreover “Cluster 1 and 2”, even though the good level of dairy efficiency, which had a positive effect animal excretion, did not achieve an efficient performance in term of nitrogen farm balance. Overall “Cluster 1” might be defined as most intensive production system because it showed typical features of intensive dairy production of Northern Italy (more arable land and more land used for growing maize, higher production level and production efficiency, higher use of concentrate and maize silage in the ration instead grass hey). “Cluster 3” was more intensive compared to “Cluster 2” but less than “Cluster 1” for that reason it could be considered representative of a medium level of intensity. “Cluster 2” was identify as less intensive production system. As it was expected farms belonged to “Cluster 1 and 2” showed an high environmental

burden on the local scale (higher nitrogen farm surplus). Analyzing the environmental impact on a global scale and in a product perspective, no relevant differences were observed among the groups and all of them should be considered at the same level (Table 7).
Only regarding total land use of "Cluster 3" showed a statistically higher value compared to the other groups. On-farm acidification and total eutrophication of "Cluster 3" with off-farm eutrophication of "Cluster 1" seemed to be higher compared to "Cluster 2", but this trend was not supported by the significance of the model. At the same level was the difference of on-farm land use between "Cluster 1" and "Cluster 3".

Tab. 7 - The effect of cluster on environmental impact per kg FPCM (Least square means - *p sistem < 0.1*).

Variable		Cluster 1	Cluster 2	Cluster 3	SE	P	P 1 vs. 2	P 1 vs. 3	P 2 vs. 3
farms	n	10	11	7					
Global warming (GWP), kg CO_2-eq	Total	1.27	1.25	1.29	0.07	0.89	0.75	0.87	0.65
	On-farm	*0.91*	*0.96*	*0.96*	*0.06*	*0.81*	*0.57*	*0.59*	*0.97*
	Off-farm	*0.36*	*0.29*	*0.33*	*0.04*	*0.34*	*0.15*	*0.57*	*0.45*
Acidification, g SO_2-eq	Total	16.0	13.9	16.8	1.23	0.15	0.14	0.63	0.07
	On-farm	*13.9*	*12.0*	*14.8*	*1.21*	*0.18*	*0.19*	*0.55*	***0.08***
	Off-farm	*2.13*	*1.86*	*1.96*	*0.30*	*0.74*	*0.44*	*0.67*	*0.78*
Eutrophication, g PO_4-eq	Total	7.82	6.86	8.12	0.59	0.20	0.17	0.70	**0.10**
	On-farm	*5.68*	*5.23*	*6.37*	*0.55*	*0.29*	*0.48*	*0.35*	*0.12*
	Off-farm	*2.14*	*1.63*	*1.76*	*0.24*	*0.18*	***0.08***	*0.23*	*0.68*
Energy use, MJ	Total	5.44	5.51	5.57	0.37	0.96	0.87	0.79	0.90
	On-farm	*4.17*	*4.98*	*4.92*	*0.43*	*0.22*	*0.11*	*0.18*	*0.92*
	Off-farm	*3.44*	*2.74*	*3.39*	*0.36*	*0.21*	*0.11*	*0.91*	*0.18*
Land use, m^2	Total	1.53	1.43	1.81	0.13	**0.09**	0.54	**0.10**	**0.03**
	On-farm	*0.64*	*0.78*	*0.93*	*0.12*	*0.17*	*0.30*	***0.06***	*0.31*
	Off-farm	*0.89*	*0.81*	*0.87*	*0.17*	*0.92*	*0.71*	*0.96*	*0.78*

4. *DISCUSSION*

4.1. Environmental impact and substances contribution

The estimated value of GWP for the production of 1 kg of FPCM is comparable to what found by Guerci et al. (2012) and in other studies (Haas et al., 2001; Thomassen et al., 2008; Müller-Lindenlauf et al. 2010). Castanheira et al. (2010) and Kristensen et al. (2011) estimated an higher contribution of on-farm activities to greenhouse gases emission compared to off-farm activities whereas O'Brien et al. (2012) and Thomassen et al. (2008) did not observed this trend very clearly. The acidification potential obtained in this study was higher compared to what found by O'Brien et al. (2012), Basset-Mens et al. (2009) and Thomassen et al. (2008) but lower than what was observed by Castanheira et al. (2010). The eutrophication potential was similar to what estimated by Castanheira et al. (2010) but higher compared to O'Brien et al. (2012) and Basset-Mens et al. (2009). The non-renewable energy use is in line to what estimated by Thomassen et al. (2008) for conventional Dutch farms nevertheless in our study the off-farm contribution was higher than the on-farm. Considering the results on land use, both the total impact and on-and off-farm contributions, were comparable with Thomassen et al. (2008) but higher than estimated in other studies (O'Brien et al., 2012; Basset-Mens et al., 2009).

Substances contribution to GWP estimated in this study were comparable with Castanheira et al. (2010) who found that CH_4, CO_2 and N_2O accounted for the 64%, 19% and 17%, respectively, of the total greenhouse gases emission. Also in several other studies was observed the high relevance of methane on the carbon footprint of milk production: de Boer (2003) reported CH_4 to be the predominant contributor to the total climate change emissions with values ranged between 48% and 65%, Basset-Mens et al. (2009) found values varied from 56% to 65%, Mc.Geugh et al., (2012) observed that around the 56.0 % of greenhouse gases emissions of the dairy farm was related to methane and in Thomassen et al. (2008) methane accounted for 34% in the conventional system and for 43% in the organic system to total climate change. Mc.Geugh et al., (2012) observed that 86% of total CH_4 was of enteric origin. Similarly, Rotz et al. (2010), Flysjö et al. (2011b), and Kristensen et al. (2011) reported the enteric fermentation as the primary source of methane (76%; 94-98%, and 85%, respectively), O'Brien et al. 2012 observed that 88.1% and 72.4% of total methane was from enteric emission for seasonal grass-based dairy farm and confinement dairy farm respectively.

As in this work, Thomassen et al. (2008) found that ammonia was the element that accounted for most of total acidification (74% in the conventional and 81% in the organic system) and Castanheira et al. (2010), observed that NH_3 emissions were responsible for 87% of the total acidification potential, whereas SO_2 and NOx have minor contributions of 9% and 4%, respectively. In the work of O'Brien et al. (2012) NH_3 volatilized during manure storage and cattle housing showed a huge variation among the systems analyzed (27.6% and 65.2% for seasonal grass-based dairy farm and confinement dairy farm respectively) the same variability was shown by NH_3 lost during manure application on farm crops (around 60% and 26% for seasonal grass-based dairy farm and confinement dairy farm respectively).

The results of eutrophication were comparable with Castanheira et al. (2010) who estimated that almost 58% of the total eutrophication potential is associated with NH_3 emissions, followed by the emission of NO_3 (35%) but in contrast with what observed by Thomassen et al. (2008) where phosphate accounted for 53% and for 31% of the impact in the conventional and organic system respectively. In the work of O'Brien et al. (2012) the nitrate losses occurred on-farm were around 90% for the seasonal grass-based dairy farm but only about the 30% for the confinement dairy farm.

Despite results from various LCA can be compared only with caution because of inevitable differences among such studies, is clear for what discuss before that enteric methane is the major driver of greenhouse gases emissions of milk production. It is, therefore, apparent that abatement of CH_4, particularly enterically derived CH_4, would result in the most significant reduction in GHG emissions (Mc.Geugh et al., 2012). At the same level, a reduction in ammonia emissions and nitrate leaching are key actions for mitigating total acidification and eutrophication potential.

4.2. Relation between farm characteristics and environmental performances

The results obtained from the PCA partly confirmed what was already highlighted in other recent studies. Regarding the animal efficiency it is well recognized enhancing the feed conversion rate of the animals to be a valid strategy in order to decrease the environmental impact per unit of product (Hermansen and Kristensen, 2011) and also a general increasing of the productivity might affect positively the environmental sustainability of the dairy farm because less animals are required to produce the same amount of milk (Capper et al., 2008). However, if breeding strategy aims to improve milk performances it might negatively affect animal health or fertility (Crosson et al., 2011). Guerci et al. (2012) showed that farming strategies based on high production intensity and high animal efficiency were better able to mitigate environmental impacts per kg of milk. Casey and Holden (2005) found a significant positive linear correlation between stocking rate and the amount of CO_2-eq. ha^{-1} but no relationship between stocking rate and GHG emissions kg^{-1} milk, Olesen et al. (2006) showed that a high N-surplus per ha was correlated with a high GHG emission per ha. Grassland seemed to have a positive effect on the environmental impact of the production system but its role was not so clear. Generally grassland needed less fertilization than arable land and that might have a positive effect on GWP, eutrophication and acidification. From the other point of view arable crops (for instance maize silage) had an higher yield per ha and required less field operations compared to grass hey production. Soussana et al. (2010) identified a strong potential of grassland C sequestration to partly mitigate the GHG balance of ruminant production systems and this was confirmed also by Rotz et al. (2010) and O'Brien et al. (2012) who observed a lower GHG emission when grassland carbon sequestration was accounted in the analysis. In this study no soil carbon sequestration were considered because the relation between livestock production and soil C sequestration is complex and uncertain (de Boer et al., 2011) more over lack of data would make difficult to estimate properly the carbon fluxes.

4.3. Farming systems and environmental performances

Many studies focused to identify the best environmental sustainable farming systems and to assess their effectiveness in mitigating the pollution burden. The results of this study did not show substantial differences between the environmental impacts of the three group of farms despite relevant differences in term of farming intensity were observed among the clusters (especially between 1 and 3 vs. 2). Intensification, defined as increased output ha^{-1}, invariably led to increased emissions when expressed on an area basis, however when expressed on a product basis, the result was less obvious (Crosson et al., 2011). Van der Werf et al. (2009) observed no difference of the environmental impacts in conventional and organic dairy systems when milk sold was considered as functional unit but, on the other hand, the conventional systems showed a significant increasing of environmental burden on land unit compared to the organic ones. Similar results were estimated by Haas et al. (2001). Müller-Lindenlauf et al. (2010) found that the climate impact of the intensive tillage based farm type was significantly lower than the climate impact of the other less intensive farm systems while the nitrate leaching potential did not differ significantly between farm groups. Oudshoorn et al. (2011) observed that no correlation between N-surplus per ha and emission of GHG per kg ECM existed. Basset-Mens et al. (2009) highlighted better environmental performances for the low input dairy system compared to more intensive systems for both product and local perspective. Also when only the environmental impact related to the product unit is considered, is complex to identify which is the best production system. Several works compared organic vs. conventional farms or grass based vs. confined farms, some of them addressed a better environmental performance to the low input systems (O'Brien et al., 2012; Belflower et al., 2012) other identify in the more intensive systems a potential reduction of the environmental pressure (Thomassen et al., 2008; Kristensen et al., 2011) other had different trends among the impact categories considered (Cederberg and Mattsson, 2000). Considering the herd size in the work of Rotz et al. (2009) large scale farms (2000 cow drylot) had lower carbon footprint per kg of milk compared to medium and small farms.

5. CONCLUSIONS

The great concern about the environmental impact of the dairy sector involved many research group to investigate which would be the best possible strategy in order to mitigate the pollution burden. There is shared consensus that improving animal and farm efficiency leads to a more environmental sustainable production. Regarding other options and technical choices their role is not so clear. This work focused to identify “farming systems” differing for the level of intensification. The definition of these systems was performed taking into account farm characteristics which were considered relevant for the environmental and technical point of view. Despite several studies highlighted that different farming systems (for instance organic vs. conventional or grass-based vs. confinement) can significantly affect the environmental burden of 1 kg of milk, the results of this work showed that when a sample of dairy farms belonging to the same production system was analyzed it was difficult to identify the environmentally “best” and “worst” farming type although relevant differences between the dairy units were observed and they make possible to distinguish two precise systems. The huge variability among the farms involved in this study is probably one of the main reasons because no statistical differences were found between the systems.

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