

## **GRAZING MANAGEMENT, AMMONIA AND NITROUS OXIDE EMISSIONS: A GENERAL VIEW**

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### **Manejo del pastoreo, emisiones de amoníaco y óxido nítrico: una visión general**

**Key words:** Ammonia, grazing, management, nitrogen, nitrous oxide, livestock.

#### **ABSTRACT**

The grazing management of grassland has a direct effect on nitrogen (N) recycling. This is an important reason why management has become an alternative to improve the grassland production and quality, in turn to make it more suitable for the environment. However, the livestock system intensification induces changes in the natural dynamics of the N cycle, accelerating gas emissions (e.g. ammonia, NH<sub>3</sub> and nitrous oxide, N<sub>2</sub>O) and leaching losses from soil under grazing. When the amount of N in the environment increases, there is an impact on smog episodes, global warming, stratospheric ozone depletion, acid rain and eutrophication of fresh water. There are different techniques to evaluate the gases emitted from the soil. This knowledge is useful to design the strategies to reduce the negative consequences of these gases on the environment. In this review, the effect of grazing managements on N gas emissions from soils and the current techniques for N gas emission measurements in the field and laboratories conditions are discussed.

**Palabras Claves:** Amoniaco, pastoreo, manejo, nitrógeno, óxido nitroso, ganadería.

## RESUMEN

El manejo del pastoreo de la pastura tiene un efecto directo en el reciclaje de nitrógeno (N). Esta es una importante razón por la cual el manejo se ha convertido en una alternativa para mejorar la producción y calidad de la pastura y a su vez hacerla más amigable con el medio ambiente. Sin embargo, la intensificación de los sistemas ganaderos inducen cambios en la dinámica natural del ciclo del N, acelerando la emisión de gases (ejemplo amoniaco, NH<sub>3</sub> y óxido nitroso, N<sub>2</sub>O) y pérdidas por lixiviación desde suelos sometidos a pastoreo. Cuando aumenta la cantidad de N en el medio ambiente, hay un impacto sobre los episodios de smog, calentamiento global, agotamiento de la capa de ozono en la estratosfera, lluvia ácida y eutroficación del agua. Existen diferentes técnicas para evaluar los gases emitidos desde el suelo. Este conocimiento es útil para diseñar estrategias para reducir las consecuencias negativas de estos gases sobre el medio ambiente. En esta revisión, se discuten el efecto del manejo del pastoreo en la emisión de gases nitrogenados desde el suelo y las técnicas actualmente utilizadas para determinar las emisiones de N gaseoso en condiciones de campo y laboratorio.

## INTRODUCTION

Cattle livestock activities generate basic products for the world's growing feeding population, but at the mean time it impacts the atmosphere, soil and water, so it becomes necessary to make these activities environmentally sound. Livestock farms use different managements to improve its profitability by means of increasing phytomass productivity, stocking rates and efficiency in the use of pastures. In many countries, pasture is one of the main sources for livestock feeding, even though, grazing generates simultaneously negatives effects.

Grazing managements, the frequency and intensity of grazing, stocking rates together with forage availability and inorganic fertilization are all aspects influencing nutrient recycling. Optimal animal nutrition guarantees the amount and the quality of grass production based on animal requirements (Gerrish, 2004). However, an incorrect management generates N losses such as NH<sub>3</sub>, N<sub>2</sub>O and leaching that would be reduced under a more efficient grazing strategy. The N cycle in grassland soils is complex because many unknown processes controlling the fate of N excess in mobile pools of soil are used (Jarvis, 1997). The

efficiency of N used in grassland depends on grazing intensity, what will be reflected in the quality and quantity of forage produced and the reduction of the N losses.

In Chile most livestock production is located in the southern regions in soils derived from recent volcanic ashes. About 5.1 millions hectares, 60% of Chilean agricultural land is derived from volcanic materials (Matus *et al.*, 2006) where livestock production is developed. In this large area, comparative studies have been carried out where N losses have been measured. However, the data amount related to N losses is scarce. Steubing *et al.* (2002) presented a range of NH<sub>3</sub> volatilization in livestock production of about 3 to 25 µg m<sup>-3</sup>. Ordóñez (2003) estimated NH<sub>3</sub> losses about 80% of organic N fertilization applied in mixed pastures. In other crops Matus (personal communication) estimated about 30% NH<sub>3</sub> emssions from the N of the fertilizer in paddy soils under field conditions. Vidal and Chamorro (2005) reported NH<sub>3</sub> emission in wheat of about 12-18 kg ha<sup>-1</sup>. Recently Núñez *et al.* (2007a, 2007b) measured a range of NH<sub>3</sub> between 31 and 41 kg ha<sup>-1</sup> using static chambers of

PVC in four grazing systems under permanent mixed pastures in a Chilean Andisol. In general, N volatilization is economically important, since it represents 13-18% of the N applied into the pastures. These losses associated to a possible environmental impact would represent a potential scaling problem that must be considered in Chilean intensive livestock production.

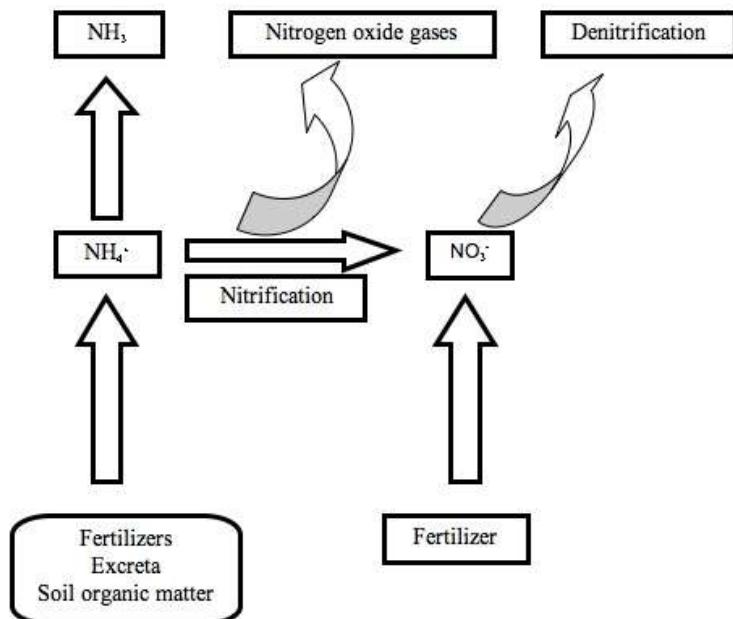
There are increasing Chilean studies regarding to N leaching in grazing pastures, but losses of about 11-29 kg N ha<sup>-1</sup> yr<sup>-1</sup> have been reported by Alfaro *et al.* (2005), 13-67 kg N ha<sup>-1</sup> yr<sup>-1</sup> in experiment fertilization with farmyard manure (Alfaro *et al.*, 2006) and 3-70 kg N ha<sup>-1</sup> yr<sup>-1</sup> in grazing systems (Alfaro *et al.*, 2007). Mora *et al.* (2007) reported about 88-90 kg N ha<sup>-1</sup> as the maximum N potential losses by leaching from sodium nitrate and urea fertilizers in irrigated mowing pasture. In general, leaching has been reported ranged from 4-90 kg N ha<sup>-1</sup> in mowing irrigated pastures of *Lolium perenne* (Ordóñez, 2003; Mora *et al.*, 2004; Mora *et al.*, 2007). Also, Demanet *et al.* (2006) reported N losses by leaching between 25 and 38 kg N ha<sup>-1</sup> yr<sup>-1</sup> for dairy grazing in Chilean Andisol. Recently Núñez *et al.* (2007b) indicated losses of about 25 and 59 kg N ha<sup>-1</sup> yr<sup>-1</sup> in grazing livestock (Holstein-Friesian). Their grazing treatments differed in frequency and intensity, so, N losses via leaching of ammonium and nitrate were determined using plot lysimeters in each of the grazed paddocks. The results mentioned above show that pastures are subject to N losses and that there is a need to collect a greater

summary data, especially for N gas losses.

This review focuses on collecting NH<sub>3</sub> and N<sub>2</sub>O data emissions from soil under grazing management systems. Some data and methodology from our own experience are presented and discussed.

### **Livestock production and environmental impact**

The N cycle is very complex, since it includes different N forms (organic and inorganic). The change of N from one pool to another in the soil is associated to the activities of microorganisms, especially the nitrifying and denitrifying bacteria. During nitrification and denitrification, different gases are produced, like NH<sub>3</sub>, N<sub>2</sub>O, NO<sub>x</sub>, NO and N<sub>2</sub> (Figure 1). In the livestock production systems managed under grazing, these gases also are produced, although some of them are not harmful, like N<sub>2</sub>; other gases are considered toxic as N<sub>2</sub>O and NH<sub>3</sub>. De Boer (2003) demonstrated that the NH<sub>3</sub> is produced in significant amounts of livestock systems under grazing (conventional and organic milk production). The main source of NH<sub>3</sub> to the atmosphere is the agriculture (Marschner, 2003; Pervanchon *et al.*, 2005). Dung and urine in this system are determinants for NH<sub>3</sub> emission as well (Ball and Keeney, 1981; Bussink, 1992; (Anderson *et al.*, 2003). It is also important to remember that NH<sub>3</sub> is a gas that produces acid rain and this phenomenon has bad effects in the soil by the acidification processes (Krupa and Moncrief, 2002; Krupa, 2003; Gerber and Menzi, 2006).



**Figure 1:** Nitrogen emissions from grassland soils, adapted from Jarvis (1997).

**Figura 1:** Emisiones de nitrógeno desde el suelo de las praderas, adaptado desde Jarvis (1997).

Other serious problems for environmental pollution is the leaching transfer to drainage (Jarvis, 1997; Ross and Jarvis, 2001a; Ross and Jarvis, 2001b) and losses by denitrification (McGechan and Topp, 2004). Also, intensive or strong animal grazing accelerates erosion (Savadogo *et al.*, 2007), accumulation of solid in the water and vegetation degradation (Savadogo *et al.*, 2007). Under such high stocking rates cattle, the soil is compacted (Russelle, 1996) specially, when vegetation is removed, resulting in a decrease of infiltration enhancing potential erosion and altering the organic and inorganic sediments and physical condition of the ecosystems (Morse, 1996; Monaghan *et al.*, 2005).

The deposition of atmospheric N, associated to an increase of agricultural and cattle activities on local and global scale,

influence significantly the chemistry of the soil, water and air N. The N cycles: past, present and future were discussed by Galloway *et al.* (2004), demonstrated that this cycle is affected by the anthropogenic activities. Imbalance of nutrients in a localized region where organic fertilizers have been applied has also been reported (Galloway, 1998). Unmanaged grasslands received most of their nitrogen reactive from biological nitrogen fixation (BNF) and atmospheric depositions; the latter source is much more important where deposition rates are large (Galloway *et al.*, 2003).

Livestock production in pastoral system influence positively some soil characteristics by reduced tillage, increasing biodiversity of microorganisms, enhancing nutrient cycling and diminishing soil erosion impact.

Under low stocking rates, nitrate pollution decreases, soil organic matter increases and physical parameters such as soil infiltration and soil aggregation stability improve (Pedraza, 1996; Hubbard *et al.*, 2004). Under high stocking rates, N could turn out into a pollutant source for drinking water, air and soil, because high N surplus, leading to N leached in the aquatic environment (Luo *et al.*, 1999; Boddey *et al.*, 2004; McGechan and Topp, 2004). Nitrogen mineralization not only involves NH<sub>3</sub> emissions (Krupa and Moncrief, 2002; Krupa, 2003; Mkhabela *et al.*, 2006), but also N<sub>2</sub>O (Mkhabela *et al.*, 2006; Takahashi, 2006; Cardenas *et al.*, 2007) and methane (Olesen *et al.*, 2006; Takahashi, 2006; Cardenas *et al.*, 2007).

Cattle grazing associated to the type of N supply (mineral or organic fertilization) increases the potentiality for N<sub>2</sub>O production in the soil (Lampe *et al.*, 2006; Granli and Böckman, 1994). Lampe *et al.* (2006) indicated that the grazing has an important effect on N<sub>2</sub>O releases since up to 57% of the accumulative N<sub>2</sub>O emission occurs during the grazing period. In grazed pastures, this potentiality is increased by the low efficiency in the N use of the bovine animals coming from the pastures, since a great percentage of the N is returned by dejection in dung and urine, and with this, N emissions from the systems increase. This gas can be produced as much by nitrification as by denitrification and this is a big challenge, that is to say, to be able to manage from grazing systems.

In temperate climates, several experiments have demonstrated that N<sub>2</sub>O emissions are higher in the winter season because the restrictive conditions of soil oxygen during grazing. The main process is produced by denitrification, where NO<sub>3</sub><sup>-</sup> is reduced to N<sub>2</sub> and in intermediate reactions gases like N<sub>2</sub>O and NO. These products contribute to global warming by absorption of radioactive energy and by degradation of the stratospheric ozone resulting in ultraviolet radiation

increase (Crutzen, 1981; Yamulki *et al.*, 2000).

### Factors affecting the potential losses of NH<sub>3</sub> and N<sub>2</sub>O in grassland

Nitrogen emissions from the soil are related to the complexity of its cycle, not only in the soil, but also in the atmosphere, water and microorganisms. Therefore, the gas released from the soil is conditioned by several factors which have incidence in the amount of the gases emitted. These depend on a complex interaction between soil properties, climatic factors, and agricultural practices (Saggar *et al.*, 2004a). Nitrogen can be lost from the pasture system through the physical processes of leaching, runoff, and erosion; the chemical process of volatilization; the biological process of denitrification; and through plant residues burning. Fluxes of NH<sub>3</sub> from the N source depend then upon environmental and soil conditions, especially pH and soil moisture.

A number of factors affect NH<sub>3</sub> emission rates (Burch and Fox, 1989) like soil pH, pH buffering capacity, cation exchange capacity (CEC), urease activity, soil moisture content, depth (Reynolds and Wold, 1987), temperature, wind velocity, relative humidity, urea particle size and application rate. In addition Barbieri and Echeverria (2003) included other factors such as organic matter and crops (amount and type of harvest remainders, source and dose of N) and Bouwmeester *et al.* (1985) included soil texture, enzymatic activity, rainfall and fertilizer management. Demeyer *et al.* (1995) and Ferguson and Kissel (1986) concluded that the NH<sub>3</sub> emission from urea application on soil surface is influenced by climatic and soil factors like water content and urease activity (Anderson *et al.*, 2003).

Organic fertilizer are also important since factors like slurry type, dry matter content, total ammoniacal N content, application

method, rate applied, slurry incorporation may affect ammonia volatilization (Søgaard *et al.*, 2002). Ammonia emissions from slurries is affected by manure composition, crop covering, weather and soil conditions (Sommer and Olesen, 1991; Bussink *et al.*, 1994; Braschkat *et al.*, 1997; Sommer and Olesen, 2000; Huijsmans *et al.*, 2001). The volatilization of NH<sub>3</sub> occurs mainly from the high concentrations of ammoniacal N that occur temporarily after the deposition of excrete or the application of slurry or fertilizer (Whitehead, 2000).

Saggar *et al.* (2004a) indicated, in addition to above-mentioned factors, the NH<sub>3</sub> release from soil to the atmosphere, depends on the NH<sub>3</sub> concentration on the soil and the partial pressure of NH<sub>3</sub> in the atmosphere. Hence, the results of NH<sub>3</sub> volatilization are negatively correlated with the percentage of clay, total N, CEC, organic carbon, hydrogen ion buffering capacity and urease activity. In a similar way, Cabezas *et al.* (1999) concluded that NH<sub>3</sub> emissions will depend on the method used in measurements.

Smits *et al.* (2003) and other researchers (Smits *et al.*, 1995; Braam *et al.*, 1997; Monteny *et al.*, 1998) indicated that the emissions of NH<sub>3</sub> from the cow house, depend on a number of factors like the diet of the cows, the design of the barn, the outdoor and indoor climate and the management of the farm, including its grazing regime. For a producer, altering the cow's diet is a relatively easy way of quickly reducing NH<sub>3</sub> emissions. This means that the NH<sub>3</sub> emissions depend on the N concentration in the food given to the cattle. Therefore, the management related to the cattle feeding, including the diet, will also influence the NH<sub>3</sub> volatilized, mainly through dung and urine. The amount of NH<sub>3</sub> that volatilizes depends also on factors such as the amount of N in the food source, size and species of the animal, housing conditions and animal waste handing practices (Anderson *et al.*, 2003).

Sommer *et al.* (2001) indicated that the volatilization was significantly affected by the amount of food given to the cows, the incidence of the solar radiation and the air temperature during measuring periods. It is important to also state that the solar radiation and rain measured one to two days before the measurement also influenced the results. That is to say, that the process of NH<sub>3</sub> volatilization from the soil does not only depend on one process, but it is also influenced by various factors. All this could increase or decrease the volume of the gas produced. The tools to manage these processes are causing environmental problems, caused by the origin of N emissions as viable and sustainable alternatives. These tools must consider the distribution pattern of the urine in patches on the grassland, and also the content of urine spilled, the distribution radio and depth of loss in certain periods of time (Koops *et al.*, 1997). Núñez *et al.* (2007a; 2007b) in pastures of the south of Chile, demonstrated that a greater grazing frequency produces larger volatilization of NH<sub>3</sub> (39.9-41.4 kg ha<sup>-1</sup> year<sup>-1</sup>), independently of the grazing intensity. Ammonia emission measurements were generated using static chambers of the same design as those used by Saggar *et al.* (2004c) nitrous oxide emissions. The concentration of ammonium-N in the acid samples was determined spectrophotometrically. Cumulative emissions for each season were obtained. Grazing intensity was measured by residual pasture post-grazing height of dry matter consumed. Thus 'heavy' grazing treatments means, a high herbage cattle consumption and the 'light' grazing treatments, a low cattle consumption. In this experiment, the heavy grazing was 10% greater than light grazing and 30% more than the control (no grazed pasture). In this study we demonstrated that the emissions of NH<sub>3</sub> vary with the grazing seasons, being higher in autumn and summer and lower in winter

and spring (Núñez *et al.*, 2007a; 2007b). In conclusion the volatilization of N is affected by the climatic conditions of the year, besides the fertilizer applied and the animal managements.

Saggar *et al.* (2004a) mentioned a number of results from investigations in different nations like England, The Netherlands, Canada, Denmark and United Kingdom, where different N volatilization sources were evaluated from different animals, like pigs, milk production, waste, dung, resulting divergent values. This could be explained

based on the conditions in which the experiments were carried out, also the N concentration in each material used, type of soil and the method used in the measurement of NH<sub>3</sub> losses.

#### **Main factors influencing NH<sub>3</sub> emissions**

A summary of the main factors that affect the NH<sub>3</sub> emissions from the soil are: pH, temperature and moisture of the soil, air temperature, radiation solar, wind speed and type of fertilizer used (Table 1).

**Table 1:** Factors that affect NH<sub>3</sub> volatilization.

**Cuadro 1:** Factores que afectan la volatilización de NH<sub>3</sub>.

Factor	Optimum for the emission	Reference
pH on the soil	Alkaline	(Ferguson <i>et al.</i> , 1984; Gezgin and Bayrakli, 1995; Roelcke <i>et al.</i> , 1996; Paramasivam and Alva, 1997; He <i>et al.</i> , 1999; Potter <i>et al.</i> , 2003; Sommer <i>et al.</i> , 2003)
Moisture of the soil	Dawn moisture of the soil	Ryan <i>et al.</i> , 1981; Bouwmeester <i>et al.</i> , 1985; Reynolds and Wold, 1987; Burch and Fox, 1989; Potter <i>et al.</i> , 2003; Beuning <i>et al.</i> , 2007)
Temperature	High	(Burch and Fox, 1989; He <i>et al.</i> , 1999; Passianoto <i>et al.</i> , 2003; Misselbrook <i>et al.</i> , 2006)
Air temperature and solar irradiation	High	(Sommer <i>et al.</i> , 2001; Misselbrook <i>et al.</i> , 2006; Beuning <i>et al.</i> , 2007)
Wind speed	Strong	(Denmead <i>et al.</i> , 1982 ; Cabrera <i>et al.</i> , 2001; Sommer <i>et al.</i> , 2001 ; Krupa and Moncrief, 2002 ; Pinder <i>et al.</i> , 2004; Beuning <i>et al.</i> , 2007)
Animal manure (type of fertilizer)	Depends on environmental and soil conditions, and of the type fertilizer	(Nathan and Malzer, 1994; Harrison and Webb, 2001; Krupa and Moncrief, 2002; Sommer <i>et al.</i> , 2004; Balsari <i>et al.</i> , 2006; Balsari <i>et al.</i> , 2007)
Other factors: dry matter content, NH <sub>4</sub> <sup>+</sup> content, length of time and time of the animals are grazing	It is very variable	(Anderson <i>et al.</i> , 2003)

**pH of the soil:** Different investigations have demonstrated that the pH of the soil plays an important role in the levels of NH<sub>3</sub> emissions (Table 1). Generally, NH<sub>3</sub> increases when soil pH is increased (Ferguson *et al.*, 1984; He *et al.*, 1999). For example, approximately 10 to 40% of N (as urea or ammonium sulfate) applied to soils (pH 4.1 to 7.7) is volatilized during the growing season, especially N from alkaline and calcareous soils (Gezgin and Bayrakll, 1995; Roelcke *et al.*, 1996; He *et al.*, 1999). In the same sense, Paramasivam and Alva (1997) in a study performed in neutral soils (pH 7.0), indicated that the pH of the soil increased (7.1-8.8) when urea was applied, which helped substantial NH<sub>3</sub> volatilization. These results match the ones reported by Potter *et al.* (2003) who found that high NH<sub>3</sub> volatilization was strongly affected by relatively high soil pH (7-9) levels.

**Temperature:** High temperature also favours NH<sub>3</sub> volatilization (He *et al.*, 1999), due to the relationship between NH<sub>3</sub> volatilization, air temperature and solar irradiation (Sommer *et al.*, 2001, Table 1). Passianoto *et al.* (2003) concluded that the emissions depend on the seasons and therefore on the changes of air and soil temperature. The same conclusions were obtained by Garcia-Montier *et al.* (2001), in the same pasture site during the dry season.

**Moisture of the soil:** Burch and Fox (1989) performed laboratory and field studies in which they measured the temperature effect and the moisture content in NH<sub>3</sub> volatilization from the soil surface where urea was applied. They concluded that under dry conditions, greater volatilization occurs in soil with greater initial moisture content, but when soil moisture conditions remained essentially constant, as occurred in the laboratory experiment, there is less volatilization from the soil when

the is more moisture content.

The temperature of the air in combination with the wind speed plays a fundamental role in NH<sub>3</sub> volatilization and the same importance has the moisture of the soil. Soil moisture content is an important factor in NH<sub>3</sub> volatilization due to its role in urea dissolution, hydrolysis and in the diffusion of urea, NH<sub>4</sub><sup>+</sup>, and NO<sub>3</sub><sup>-</sup> in the soil. Burch and Fox (1989) concluded that when moisture in the flow-through experiment increases, less volatilization occurs in wetter soil, possibly due to greater nitrification in the wetter soil, which reduces NH<sub>4</sub><sup>+</sup> concentration levels and allows less volatilization. Assuming adequate moisture levels, higher temperature produces greater NH<sub>3</sub> losses, due presumably to an increase in the urease activity and possibly to evaporation driven concentration of ammoniacal N near the surface.

Potter *et al.* (2003) concluded that soil-wetting patterns can strongly influence NH<sub>3</sub> emissions. Generally, moistered soils emit less NH<sub>3</sub> than drier soils, due to the high solubility of NH<sub>3</sub> and the lower gas diffusivity in wetter soils. The explanation of the process under dry conditions is that urea hydrolysis had ceased within 24 h after the application initial moisture content, resulting in only 25% of the urea being hydrolyzed. In soils with 0.25% as initial moisture content, hydrolysis continued for at least 3 d, with a total of approximately 80% being hydrolyzed at the end of day 7 (Burch and Fox, 1989). Reynolds and Wold (1987) concluded that a variation in the moisture percentage (10 to 30%) with an application of ammonium sulphate fertilizer, affects the level of NH<sub>3</sub> emissions. Barrington *et al.* (2002) found that the losses of N by volatilization were not affected by the moisture content or aeration regimen. However, the results shown previously, demonstrate precisely the opposite.

Volatilization of  $\text{NH}_3$  is related to soil moisture content, with maximum losses occurring when fertilizers are applied to soils at or near field capacity (Ryan *et al.*, 1981). The same happens with urea fertilizer ( $\text{NH}_2\text{CO}$ ), that depends on the soil moisture, time and amount of precipitation after fertilizer application per season and also on the fertilizer applied (Craig and Wollum, 1982). In that same sense, Bouwmeester *et al.* (1985) indicated that N losses increased 8 to 21% when initial soil moisture was increased.

**Wind speed:** Wind speed has an effect in  $\text{NH}_3$  volatilization and Cabrera *et al.* (2001) demonstrated that wind speed influences  $\text{NH}_3$  emissions. With wind speed of 1 cm above the soil surface varied between 0.07 and 0.47  $\text{m s}^{-1}$  and remained above the threshold of 0.135  $\text{m s}^{-1}$  approximately 88% of the time, getting losses in a range of 11.7-12.1%. The influence of high winds on volatilization was particularly strong in the small height crop. The greater concentrations of volatilized N happened in a range of 6.6 to 7.2  $\text{m s}^{-1}$  in a concentration of 2.06 to 2.00  $\text{kg N ha}^{-1} \text{hour}^{-1}$ , whereas the lowest volatilization was of 0.05  $\text{kg N ha}^{-1} \text{hour}^{-1}$  in 2.2  $\text{m s}^{-1}$  of wind speed (Denmead *et al.*, 1982).

#### **Controlling factors of $\text{N}_2\text{O}$ emission**

Oenema and Sapek (2000) indicated that two categories of factors control  $\text{N}_2\text{O}$  emissions, i.e., environmental factors and management factors. Soil factors such as  $\text{NO}_3^-$ -N and  $\text{NH}_4^+$ -N, aeration, organic matter and soil water content are the main factors. In climate factors, precipitation and temperature are important ones. In fact, this emission depends on the balance between  $\text{N}_2\text{O}$  production from nitrification and denitrification which depends, at the same time, on the soil moisture content (Abbas and Adams, 1998). Denitrification is a process where  $\text{NO}_3^-$  is consumed and it is considered as a harmful process, since it reduces the

content of available  $\text{NO}_3^-$  for plants. The specie  $\text{N}_2\text{O}$  is produced by nitrification and denitrification, and both are carried out by microorganisms, mainly by bacteria (Wrage *et al.*, 2001; Wrage *et al.*, 2004). The main important management factors in grassland systems on  $\text{N}_2\text{O}$  emission are (Oenema and Sapek, 2000): i) nitrogen fertilizer, manure application and timing of application, ii) the intensity of grazing, iii) soil compactation (Bhandral *et al.*, 2003; Bhandral *et al.*, 2007) and grassland reseeding. Pinto *et al.* (2004) demonstrated that soil tillage increases  $\text{N}_2\text{O}$  emissions in perennial pasture, iv) drainage and irrigation must also be considered and v) liming application. Grazing is important because it determines how much dung and urine is deposited on grassland from the animals. Therefore, changes in current managements can decrease  $\text{N}_2\text{O}$  emissions (Velthof *et al.*, 1996; Velthof and Oenema, 1997). However, nitrogen fertilization will generally stimulate the production of  $\text{N}_2\text{O}$  and denitrification (Corré *et al.*, 2000).

#### **Losses of nitrogen associated with animal manure**

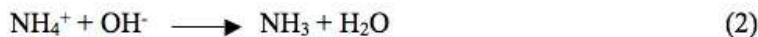
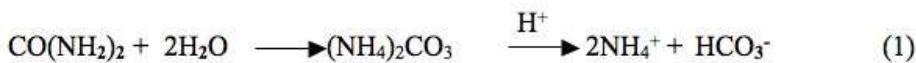
Ammonia volatilization increase with high temperatures, pH, and droughts. Applying manure when temperatures are below freezing can also increase  $\text{NH}_3$  emissions. In the case of manure application with urea, the losses of  $\text{NH}_3$  increase with soil temperature and wind speed but this is suppressed by relative humidity (Nathan and Malzer, 1994). As mentioned before, Smits *et al.* (2003) indicated that  $\text{NH}_3$  emissions from a cow house depend on a number of factors like the cows' diet, the climate and the management of the farm.

Huijsmans *et al.* (2003) reported that  $\text{NH}_3$  volatilization from field-applied manure is affected by weather conditions, manure characteristics, soil conditions, crop covering and method of application or incorporation of N. Nitrogen losses as  $\text{NH}_3$  in dairy factory

effluent irrigation are very small (Cameron *et al.*, 2002). Anderson *et al.* (2003) showed that the major factors that influence NH<sub>3</sub> emissions from livestock depend on production steps.

Volatileization will depend on grazing managements and this is indispensable to reduce the emissions in intensive production systems. The strategies consider four great steps in which NH<sub>3</sub> is produced: animal

confinements, manures preadings and animal grazing. Nitrogen in the urine is in the form of urea CO(NH<sub>2</sub>)<sub>2</sub> which is rapidly hydrolyzed to form ammonium carbonate (Anderson *et al.*, 2003; Bolan *et al.*, 2004). Hydrolysis is facilitated by the urease enzyme, which is abundant in the soil and plant roots, as well as in animal dung (Jarvis and Pain, 1990; Whitehead, 1995), as shown in reactions 1, 2:



### **Ammonia and nitrous oxide emissions in grasslands**

Livestock productions are very vulnerable to N losses, especially in the form of gases like NH<sub>3</sub> and N<sub>2</sub>O. These losses can reach levels between 20-30% of the N that enters (inputs) to the system (Bouwman *et al.*, 2005). In the case of NH<sub>3</sub> losses, it can represent a) between 8-9% of the input in temperates regions and between 12-21% in tropical regions (Bouwman *et al.*, 2005). In Chile it represents about 13-18% in sourthern regions (Núñez *et al.*, 2007a, 2007b) and about 10% of N applied like urea fertilizer (Salazar *et al.*, 2007), that which implies an important loss of this nutrient by gaseous way. Ammonia volatilization in livestock systems under grazing is very variable, since

these losses depend on many factors (Table 1). Table 2 shows NH<sub>3</sub> emissions fluctuate in a range of 2-204 kg ha<sup>-1</sup> yr<sup>-1</sup>, however the average range in countries like New Zealand, England and Australia is between 40-50 kg ha<sup>-1</sup> yr<sup>-1</sup>. These emissions are high from an economic and environmental point of view. The level of the emissions generally varies with the type of livestock production (dairy, beef, build, and country side), amount and type of animal's soil pH, among others (Table 1 and 2). In general, dairy livestock systems and intensive systems have a higher emission, but these also have high inputs of N and NH<sub>3</sub> volatilization increase.

**Table 2:** Ammonia volatilization ( $\text{kg N ha}^{-1} \text{ year}^{-1}$ ) from grazing systems according to several researchers.**Cuadro 2:** Volatilización de amoníaco ( $\text{kg ha}^{-1} \text{ año}^{-1}$ ) desde sistemas de pastoreo de acuerdo a diferentes investigadores.

<b>Country</b>	<b>Production Systems</b>	<b>NH<sub>3</sub></b> <b>Volatilization (kg ha<sup>-1</sup>)</b>	<b>References</b>
New Zealand	Cattle grazing	15-204	Ledgard <i>et al.</i> (1999)
New Zealand	Dairy farms	15-65	Ledgard <i>et al.</i> (1998)
New Zealand	Grazed swards (dairy farm)	14	Jarvis and Ledgard (2002)
Sao Pablo, Brazil	Intensive grazing	2-13	Martha <i>et al.</i> (2003)
Germany	Permanent grazing grassland	1.7-4.9	Lampe <i>et al.</i> (2004)
England and Wales	Cattle grazing	24-30 soil sandy loam	Webb <i>et al.</i> (2005a)
England and Wales	Cattle grazing	17-24 soil clay loam	Webb <i>et al.</i> (2005a)
United Kingdom	Grazed swards (dairy farm)	8.3	Jarvis and Ledgard (2002)
United Kingdom	Beef cattle in pasture	3-80	Oenema (2006)
United Kingdom	Dairy cattle intensive grazing	80	Oenema (2006)
SW England	Dairy farmer	10	Jarvis (1993)
British	Grassland	30-40*	Brown <i>et al.</i> (2005)
The Netherlands	Dairy cattle intensive	32-129	Oenema (2006)
Netherland	Dairy farming system (manure fertilizer)	30	Aarts <i>et al.</i> (2000)
Netherlands	Grassland grazed by dairy cattle	39-42	Bussink (1992)
Western Jutland	Grazing systems	115 (0.07-2.1 kg ha <sup>-1</sup> d <sup>-1</sup> )	Sommer <i>et al.</i> (2001)
Edinburgh	Manager grassland with cattle	1.8-2.3	Di Marco <i>et al.</i> (2004)
Gippsland, Australia	Temperate perennial pasture	19-43	Eckard <i>et al.</i> (2003)
Swiss Central Plateau	Dairy production	7-29	Dueri <i>et al.</i> (2007)
Argentina	Fertilizer pasture	17-48	Barbieri and Echeverria (2003)
Chile	Grazing systems	31-41	Núñez <i>et al.</i> (2007a; 2007b)
Chile	Grazing systems under urea fertilizer	10	Salazar <i>et al.</i> (2007)
Chile	<i>Lolium perenne</i> pasture	80%*	Ordoñez (2003)

\*Estimated.

At global levels,  $\text{N}_2\text{O}$  emissions are much lower than  $\text{NH}_3$  emissions (Tabla 3). The results show values between  $0.03-107 \text{ kg ha}^{-1} \text{ yr}^{-1}$ , however the tendency is to find a production of inferior levels than  $10 \text{ kg ha}^{-1}$ . High values of emissions presented in Table 3 correspond to N gas emissions by denitrification and therefore other gases are included.

These explain the high values for example from Jarvis (1993), Ledgard *et al.* (1998, 1999), Rees *et al.* (2004) and Dueri *et al.* (2007). The emissions of this gas in the livestock systems depend on environmental factors during the season, any changes in the scenario it will produce a differential emission from the previous one (van Groenigen *et al.*, 2005).

**Table 3:** Nitrous oxide ( $\text{N}_2\text{O}$ ) emission in several countries of the world.**Cuadro 3:** Emisión de óxido nitroso ( $\text{N}_2\text{O}$ ) en diferentes países del mundo.

Country	Production system	$\text{N}_2\text{O}$ emisión ( $\text{kg ha}^{-1} \text{yr}^{-1}$ )	References
United Kingdom	Grassland	0.86 $\mu\text{g kg}^{-1} \text{ha}^{-1}$	Abbasi and Adams (1998)
United Kingdom	Grassland	4.7 $\mu\text{g kg}^{-1} \text{ha}^{-1}$	
United Kingdom	Agriculture	1.01-3.22	Brown <i>et al.</i> (2002)
Wales, UK	Grassland soil under field condition	0.1-5.28	Abbasi and Adams (2000)
SW England	Dairy farmer	42-48*	Jarvis (1993)
England and Wales	Grazing grassland	1-4	Webb <i>et al.</i> (2005a)
United States	Different system production	0.1-1.0	Burke <i>et al.</i> (2002)
	Rangeland dry land	0.4- 2	
	Irrigated system	3-6	
Europe	Livestock dairy systems	0.9	Schils <i>et al.</i> (2005)
Mongolia	Grassland	0.06-0.21	Wang <i>et al.</i> (2005)
France	Extensive grassland	1.9- 2.9	Pervanchon <i>et al.</i> (2005)
	Intensive grassland	6.6-9.9	
	Extensive grazed grassland	1.4-2.0	
	Intensive grazed grassland	6.2-9.4	
China	Agricultural soil	9.2-11.7 $\mu\text{g N kg}^{-1} \text{0 and 14.4-248.8 } \mu\text{g N kg}^{-1}$ (soil group 0 and 1)	Cheng <i>et al.</i> (2005)
New Zealand	Cattle grazing	7-30*	Ledgard <i>et al.</i> (1999)
New Zealand	Dairy farms	5-25*	Ledgard <i>et al.</i> (1998)
New Zealand	Grazed grassland (grazing regimen)	4-8	Saggar <i>et al.</i> (2007)
New Zealand	Dairy-grazed pasture	26.4-32 $\text{g ha}^{-1} \text{d}^{-1}$	Saggar <i>et al.</i> (2004c)
New Zealand	Restricted autumn dairy grazing	0.6-1.4	de Klein <i>et al.</i> (2006)
Edinburgh-Scotland	Grassland	7-9 (1999)* and 4-107 (2002)*	Rees <i>et al.</i> (2004)
Australia	Intensive managed irrigated pasture	4.5-5.5**	Phillips <i>et al.</i> (2007)
Australia	Intensive managed irrigated pasture	5-6 (NGGI)*** 14-17 (IPCC)***	Phillips <i>et al.</i> (2007)
Australia	Intensive pasture systems	0.3-13.3	Eckard <i>et al.</i> (2006)
Gippsland, Australia	Temperate perennial pasture	5-10*	Eckard <i>et al.</i> (2003)
Spain	Tillage of perennial pasture	0.027-0.56	Pinto <i>et al.</i> (2004)
Swiss Plateau	Central Dairy production	6-32*	Dueri <i>et al.</i> (2007)
Ireland	Fertilized grazed	2.2-8.3 (year 1) 13-24 (year 2)	Hyde <i>et al.</i> (2006)
Chile	Grazing systems	2.8-3.2***	Núñez <i>et al.</i> (2007a; 2007b)

\*Denitrification; \*\*Measured; \*\*\*Estimated.

## Measurement technique of NH<sub>3</sub> and N<sub>2</sub>O

### *Ammonia emissions*

Different techniques and approaches have been used. In this review, techniques such as 1) micrometeorological sampler (FAO, 2001; Cabrera *et al.*, 2001; Kissel *et al.*, 2004), 2) forced draft system (Hargrove *et al.*, 1987; Buresh, 1987; Roelck *et al.*, 1996), 3) chamber and wind tunnel (Cabrera *et al.*, 2001; Chantigny *et al.*, 2004), 4) indirect measurement and <sup>15</sup>N (Craig and Wollum, 1982; Hargrove *et al.* 1987) and 5) Other method will be discussed (Denmead *et al.*, 1982; Fillery and de Datta, 1986; Hargrove *et al.*, 1987; Fox *et al.*, 1996; Cabrera *et al.*, 2001; Sakurai *et al.*, 2003; Saggard *et al.*, 2004a).

**Micrometeorological sampler:** The micrometeorological techniques (MT) consist in the use of analysis of the atmospheric concentration of the gas and meteorological measurements such as wind speed, wet dry-bulb air temperature, net radiation, and heat fluxes. These techniques do not disturb the environment conditions (FAO, 2001). The MT are typically considered the most accurate, but due to the large plots required they are not practical for simultaneous evaluations of several treatments (Cabrera *et al.*, 2001; Kissel *et al.*, 2004).

**Forced-draft system:** The forced-draft system is a method that utilizes a dynamic chamber system for NH<sub>3</sub> measurements (Hargrove *et al.*, 1987). Buresh (1987) in laboratory studies demonstrated that forced-draft system can be applied to laboratory. In the experiment the influence of soil properties on ammonia emissions and N loss from different sources of fertilizer were determined. The results obtained with this type of techniques are more precise. This method has been applied for studies

of ammonia emissions in soils columns in laboratory. Roelck *et al.* (1996), obtained accurate results and adapted in calcareous soils of China. This method can also be applied to the field conditions. It has the advantage of direct measurements of volatilized NH<sub>3</sub>, while minimizing the disturbance of the environmental field. However, few methods are available for precise measurements of NH<sub>3</sub> volatilized in the laboratory. The method mentioned has been proposed for the measurement of NH<sub>3</sub> emissions in the field.

**Chamber and wind tunnel:** The chamber and wind tunnel uses, small structures are required for the establishment of the system and also penetrations in the soil, these also have an environmental potentiality to alterate the conditions (Cabrera *et al.*, 2001). Enclosure or chamber techniques involve the use of cuvettes, chambers or boxes placed over the bare soil surface with low vegetation cover (FAO, 2001). On the other hand, the wind tunnel method consists of an inverted acrylic plastic box connected to a steel duct housing a fan (Chantigny *et al.*, 2004) which allows its application. The wind tunnels has been used in different investigations with excellent results (Bouwmeester *et al.*, 1985; Thompson *et al.*, 1990; Sommer and Olesen, 1991; Cabrera *et al.*, 1993; Cabezas *et al.*, 1999; Chantigny *et al.*, 2004). In Chile (Núñez *et al.*, 2007a, 2007b), reported the use the chamber of PVC to measure NH<sub>3</sub> volatilization in pasture under grazing and with low level of vegetation and soil disturbance. In paddy soils chambers of PVC have been used in which rice plants were enclosed and the air fluxes were trapped with sulfuric acid at the outlet of PCV chamber (Matus, personal communication).

**Indirect measurement and <sup>15</sup>N:** These techniques are used for the NH<sub>3</sub> measurements. For example indirect <sup>15</sup>N

balance and direct method with a semi-open static ammonia collector, are also used. Hargrove *et al.* (1987) concluded that the efficiency of the semi-open static collector in retaining NH<sub>3</sub> was correlated with <sup>15</sup>N balance method. Craig and Wollum (1982) used an indirect method with topless plastic boxes and the concentration of NH<sub>3</sub> was determined afterwards in the laboratory.

**Other method and techniques for NH<sub>3</sub> measurement :** Saggar *et al.* (2004a) indicated the existence of techniques for NH<sub>3</sub> measurements, such as, mass balance micrometeorology (Pain *et al.*, 1989), theoretical profile shape (Gordon *et al.*, 2000), direct manure sampling (Lauer *et al.*, 1976), chemical trapping with volatilization chambers (Hoff *et al.*, 1981; Ferguson and Kissel, 1986). These methods have demonstrated certain levels of precision in the determination of NH<sub>3</sub> emissions, even though they have high application costs.

Sommer *et al.* (2001) studied the NH<sub>3</sub> volatilization with the atmospheric mass balance technique and the spatial variation with a dynamic chamber which was used by Doorn *et al.* (2002) and Barrington *et al.* (2002). In the same way Cabrera *et al.* (2001) used the same methodology, obtaining satisfactory results. The calculation of NH<sub>3</sub> emissions from domestic animal's waste is based upon a mass balance method that uses average N excretion from different domestic animal categories and subsequent NH<sub>3</sub> losses during housing storage and land application under grazing (Doorn *et al.*, 2002).

Scholtens *et al.* (2003) used the tubular glass chambers to measure passive ammonia flux samplers, this method is applicable to naturally ventilated animal houses as well as to fan ventilated. Ratio method is also used in the measurement of NH<sub>3</sub> (Caldwell *et al.*, 2002). This method was used in forest crops due to its easy application. The gas chromatograph-photoionization detector is another method to study NH<sub>3</sub> in the air (Yamamoto *et al.*, 1994; Phillips *et al.*, 2001).

Fox *et al.* (1996) referred to various researchers who have used the static trap method to measure NH<sub>3</sub>, which is generally believed to be inaccurate because of its marked effect on soil and to air properties that control the rate of NH<sub>3</sub> volatilization losses. Mount *et al.* (2002) mentioned other methods such as photofragmentation, laser induced fluorescence, denuders, citric acid coated filters, chemiluminescence, Fourier and other spectroscopic techniques. These techniques are recommended for the direct spectroscopic detection of NH<sub>3</sub> without the intervention of collecting medium for unequivocal continuous readings and calibrated measurements.

Other method used in the grassland, is the ammonia sensorial semiconductor. This method measures the NH<sub>3</sub> concentration in situ, which changes from one moment to another (Kawashima and Yonemura, 2001). This conventional method lacks of simplicity and convenience because it requires an electric power source. Another method proposed by Clough *et al.* (2003) is the use of ion flow tube mass spectrometry to capture gas and read it by chromatography. With this method excellent results have been obtained in the measurement of NH<sub>3</sub>.

#### **Nitrous oxide emissions** □

It is recommended to use the chamber (Flechard *et al.*, 2007; Allard *et al.*, 2007), to establish the N<sub>2</sub>O production and concentration through denitrification, using acetylene. The sample of this gas is analyzed with a gas chromatograph afterwards (Hackl *et al.*, 2000; Flechard *et al.*, 2007; Allard *et al.*, 2007). Saggar *et al.* (2004b; 2004c), Pinto *et al.* (2004) and Carter (2007) used the chamber for N<sub>2</sub>O measurements in grazed and ungrazed areas and determined the N<sub>2</sub>O in the laboratory by another technique, with a good level of accuracy. For example, Saggar *et al.* (2004b; 2004c) used for the determination of N<sub>2</sub>O gas chromatography and other researchers have also used this

methodology with reliable and precise results (Choudhary *et al.*, 2001; Choudhary *et al.*, 2002; Bhandral *et al.*, 2003; Bhandral *et al.*, 2004; Bolan *et al.*, 2004).

A very used method for determinating N<sub>2</sub>O emissions is the application of estimation factors like (IPCC), that is a very simple methodology and that has been applied to the N<sub>2</sub>O emissions from grassland managements with manure fertilization (IPCC, 1996; Dämmgen and Grunhage, 2002).

Bolan *et al.* (2004) enumerated several methods to study denitrification: (acetylene (C<sub>2</sub>H<sub>2</sub>), <sup>15</sup>N labeled fertilizer, the use of the radioactive isotope <sup>15</sup>N, measuring N<sub>2</sub> and N<sub>2</sub>O directly, use of helium and gas chromatography (Clough *et al.*, 2003) and electron capture detector (Hatch *et al.*, 2005). Phillips *et al.* (2007) used the micrometeorological techniques, while Cardenas *et al.* (2003) has obtained excellent results using an automated laboratory incubation system. Another used technique is isotope analysis, using closed-chamber and the gas samples analyzed by mass spectrometer coupled to a gas chromatograph unit (Lampe *et al.*, 2006; Cardenas *et al.*, 2007; do Carmo *et al.*, 2007, Liu *et al.*, 2007).

#### Pasture management practices for efficient nitrogen cyclins

Soil compaction is one of the major problems facing modern agriculture and some factors that may cause it are: an over use of intensive cropping machinery, short crop rotations, intensive grazing and an inappropriate soil management. The compaction of soil is an essential factor in N losses because of its great importance on the grazed area. Therefore, excessive stocking rates increase compaction, favouring the anaerobic conditions of the soil and increasing N<sub>2</sub>O emissions. Hamza and Anderson (2005) indicated various management activities of the grassland and

of the stocking rates that benefit the soil, with effects in the increment of the water infiltration in the soil (Li *et al.*, 2001) and minimization in N losses, e.g., of N<sub>2</sub>O (Ball *et al.*, 1999b).

Management of the stocking rates in the grazing systems, the reduction of N<sub>2</sub>O emissions are directly influenced and also affects the chemical, biological and physical properties of the soil. For example, Bhandral *et al.* (2003) obtained an increment in the N<sub>2</sub>O emissions as a result of soil compaction. When comparing the emission of N<sub>2</sub>O originated from compacted and non compacted soil, significant differences were found, depending on the type of fertilizer treatment applied. Nevertheless, the greater emission from compacted soil took place when NO<sub>3</sub><sup>-</sup> was applied as potassium nitrate, where a loss of 61-74 kg N ha<sup>-1</sup> occurred, which is superior in more than 50%, in general, from all the treatments applied (urine, ammonium sulphate and urea). In uncompacted soil, the losses were reduced to 4.37 kg N ha<sup>-1</sup>, due to the management.

A technique to reduce NH<sub>3</sub> emissions is through improving mineral management, since reducing the N amount per hectare applied. Burke *et al.* (2002) concluded that the volatilization produced is the result of the management implemented in the production system, since many practices of land management include grazing of the livestock, burning of tall grass pasture and cropping, all of which increase N volatilization.

In the case of soil highly calcareous, a useful management is the application of urea phosphate fertilizer, with the aim of reducing NH<sub>3</sub> emissions in comparison with the use of only urea (Stumpe *et al.*, 1984). In terms of fertilization, Fan and Mackenzie (1993) expressed that the reduction of NH<sub>3</sub> losses may be obtained by: (i) coating the urea granulates with materials that urea shows the dissolution of (ii) reducing hydrolysis

with urease inhibitors, (iii) adding neutral salts containing Ca or K and (iv) reducing microsite pH with acidic materials. In an investigation carried out by Fenn *et al.* (1982), the results showed that the use of  $\text{NH}_4\text{NO}_3$ ,  $\text{NH}_4\text{Cl}$  and  $\text{NH}_4\text{H}_2\text{PO}_4$  with urea or  $(\text{NH}_4)_2\text{SO}_4$  reduced  $\text{NH}_3$  emissions. A strategy to reduce  $\text{NH}_3$  emissions is to mix acid fertilizers like  $\text{H}_3\text{PO}_4$ ,  $\text{HNO}_3$ , and  $\text{HCl}$  with urea. Being urea the most used fertilizer in grasslands, due to its low cost and N concentration.

Chen *et al.* (2004) suggested several mechanisms to reduce N gas emissions from the soil to the atmosphere. Among these strategies is the increase of N utilization by animals through crude protein and metabolizable energy in forages and improving perennial plant N uptake of mineral soil through developing and maintaining balanced legume/grass pastures and using deep-rooted grasses. In a similar way, Smits *et al.* (2003) showed that nutritional measures related to  $\text{NH}_3$  emissions should focus on reducing urinary urea concentration by reducing the N in the diet (Frank *et al.*, 2002). Another mechanism to reduce  $\text{NH}_3$  emissions is to increase the time the cattle spends grazing (Webb *et al.*, 2005a).

Other mechanisms including the use of urease inhibitors such as phenyl phosphate diamidate, hidroxamate and benzoquinone, reduce the hydrolysis of the urea and can also reduce the volatilization of ammonium. Another technique is to combine the urea with other compounds like aldehyde and sulphide or to use urea in foliate fertilizers (Prasad, 1998). Other mechanisms reduce the quantity of urea in the animal's urine depending on the diet supply (Russelle, 1996; Soren *et al.*, 1998). In the case of fertilization with waste or animal liquid manure, it is recommendable the application by injection that introduces the fertilizer inside the soil with the purpose of reducing gas emissions (Thompson *et al.*, 1987; Misselbrook *et al.*,

2002; Huijsmans *et al.*, 2003; Rodhe *et al.*, 2004). Other managements include the dilution of the manures, separation by type of manure, water application after the distribution, application of the organic material in the appropriate epoch (season, day and time) of agreement with Webb *et al.* (2005b).

A strategy used with high efficiency by the farmers is to remove the excrete in autumn from the grassland (Russelle, 1996). Recently, other techniques suggested, consisted on adding to the animal's diet adipic acid to 1% with the purpose of reducing the pH of the urine and thus the emission of  $\text{NH}_3$  (van Kempen, 2001).

Where there is heavy grazing, fertilizers are being applied, and exotic species are being established and the native grassland diversity is greatly impacted. McIntyre *et al.* (2003) commented and compared the results found with those obtained in different periods by Whalley *et al.* (1978), Prober and Thiele, (1995) and concluded that the stability of the grassland is altered.

Sustainability of agricultural systems depends, to a great extent, on the maintenance of soil properties within levels of variation that would allow their restoration and would not affect either crop production or the environment (Studdert *et al.*, 1997). For example, changes in species composition due to land use, grassland management or other environmental changes impact on the N cycle and Hooper and Vitousek (1998) concluded the same.

Different grazing systems exist, for example the type not regulated, that is to say, a grazing where there is no control over animals in the pasture and this has direct effects on the structure of the pasture and input-output of N. Singer and Schoenecker (2003) indicated that not regulate grazing may alter any components of the N cycle, like N fluxes on an annual basis into and out of the ecosystems ( $\text{NH}_3$  volatilization from ungulate urine, denitrification,

$\text{NO}_3^-$  leaching, N losses via wind and surface runoff erosion, spatial movement).

Agriculture is based on the knowledge of the effect of management practices on soil properties and how they affect soil-crop relationships (Francis and Clegg, 1990; Studdert *et al.*, 1997). It is necessary to focus on N management strategies of farms in order to utilize biological and technological means to reduce N losses.

Many actions can be determined to reduce the environmental impact of N from dairy production systems (Misselbrook *et al.*, 2006; Webb *et al.*, 2006). For example, it is essential to consider the soil type, crops, animals, feeding, housing and manure managements (Børsting *et al.*, 2003). Diet composition affects the ratio between urinary and dung N, as well as the composition of both urinary and dung N compounds. Therefore, a form of reducing N losses in the systems under grazing, is controlling the application dose and time of N in order to reduce the amount of N in the forage and therefore in the dung and urine of the animal.

Regarding  $\text{NH}_3$ , Smits *et al.* (2003) reported that the emissions depended on several aspects of farm management, like the intensity of land use, grazing management, dietary balance and slurry application. The application of a model of field observation to measure emissions from the different sources at several commercial farms showed that  $\text{NH}_3$  emission was high ( $\text{kg N ha}^{-1}$ ). There are large variations in total calculated  $\text{NH}_3$  emissions and these ranged between 27 and 61  $\text{kg N ha}^{-1} \text{ yr}^{-1}$  with an average of 48  $\text{kg N ha}^{-1}$ . They concluded that different sources that were contributing to the total losses by emission, among which are the cows, contributed with 30 and 57% (average 47%) of the total of emissions per hectare. Storage contributed from 0 to 11% (average 1.5%), grazing from 1.4 to 15% (average 8.4%), slurry application from 21 to 49%, with an average of 34%,

and fertilizer with 0 and 14% (average 9.3%). The study demonstrated that the main factor that was contributing mostly to the emissions was the cowshed.

Haas *et al.* (2001) demonstrated that the management system of the grazing has different categories of impact in the animals used, the pasture, the soil, the cycle of nutrients, surface water, the amount of energy used, as well as an impact in the surroundings and other ecosystems. For example, problems of acidification and eutrophication are mentioned. Nevertheless, the extensified system is different from the intensified and organic. Organic agriculture has shown inherent ecological advantages in the production systems, which most indicators were significantly compared with intensive farming and other systems. There is a greater N fertilizer use and therefore a greater risk of contamination.

When using in an efficient way the application of urine in the pasture, we are implementing management practices to improve the quality of the grassland and to be successful in the use of urine, it is necessary to consider that the application of urine onto the soil often results in instantaneous N emissions (Koops *et al.*, 1997). On the other hand, Monaghan and Barraclough (1993) found high emissions of  $\text{N}_2$  but not of  $\text{N}_2\text{O}$  immediately after the urine application. Nevertheless, Sherlock and Goh (1983) indicated that they found  $\text{N}_2\text{O}$  emissions immediately after applications. In relation to this, many researchers conclude that the humidity of the soil is the factor that is controlling the emission of  $\text{N}_2\text{O}$  (Davidson, 1992; Mumey *et al.*, 1994; Koops *et al.*, 1997; Glass, 2003). According to de Klein *et al.* (2001), the different strategies of management (Table 4) should consider the N cycle in agricultural systems in order to reduce  $\text{N}_2\text{O}$  emissions.

**Table 4:** Management options to reduce N<sub>2</sub>O emissions in agricultural systems (de Klein *et al.*, 2001).**Cuadro 4:** Opciones de manejo para reducir las emisiones de N<sub>2</sub>O en sistemas Agrícolas (de Klein *et al.*, 2001).

<b>Recommendations</b>
<i>Match N supply with crop demand</i>
Use soil/ plant testing to determine fertilizer needs
Limit N mineralization by minimizing fallow periods
Optimize split application schemes
Match N application to reduced production goals
<i>Tighten the N cycle</i>
Integrate animal and crop production systems by reusing manure in plant production
Maintain residue N on production site
<i>Use advanced fertilization techniques</i>
Use controlled release fertilizers and nitrification inhibitors
Place fertilizer below surface
Use foliar feed fertilizers
Match fertilizer type to seasonal conditions
<i>Optimize tillage, irrigation and drainage</i>

On the other hand Saggar *et al.* (2004a) stated many numbers of practices to reduce the gas discharges of N from the soil. For example, incorporating slurry into the soil may reduce NH<sub>3</sub> emissions, but such slurry applications may enhance N<sub>2</sub>O emissions. Similarly, anaerobic digestion can reduce N<sub>2</sub>O emissions from the land application of effluents, but there may be a potentiality for higher NH<sub>3</sub> emissions resulting from a higher

substrate pH, and also for higher emissions of CH<sub>4</sub>. The strategies for reducing N gas emissions from grazed pastures should focus on reducing the gases emitted from animal excreta. Another mitigation option would be to manipulate the amount of N offered to the animal to reduce N excretion and thus reduce the NH<sub>3</sub> emissions (Smits *et al.*, 2003).

### Grazing management practices and nitrogen emissions ( $\text{NH}_3$ AND $\text{N}_2\text{O}$ )

Vallentine (2001) referred to the grazing intensity, to the demanded amount of animal forage placed on the sowing crop forage and to the resulting level of defoliation happening during grazing. Diverse investigations classify the intensity of the pasturing in the levels of; heavy grazing, moderate grazing, and light grazing. However, other names are used to refer to the intensity of the grazing like overgrazing, proper grazing, and undergrazing. In this review we will only refer in relation to the intensity and management of the grazing and N emissions from the grassland.

It is accepted that intensive animal management has been responsible for increases in atmospheric  $\text{NH}_3$ /ammonium ( $\text{NH}_4^+$ ) concentrations (Pain *et al.*, 1998), because a high stocking rates causes high  $\text{NH}_3$  emissions (Børsting *et al.*, 2003) due to a high forage nutrient concentration (Ivanova-Peneva *et al.*, 2006). Changing from urea to another form of N fertilizer would reduce emissions from  $>8$  to  $<1 \text{ kg ha}^{-1}$  (an overall reduction of 26%), according to Jarvis and Ledgard (2002).

The livestock system is characterized by a combination of plant and animal production (Børsting *et al.*, 2003). Grazing managed pasture systems are the main systems for livestock production (i.e. sheep, beef and dairy cattle, and deer) in many countries (Bolan *et al.*, 2004). In grazed pastures, N is derived from biological fixation of atmospheric N, through the addition of manures and fertilizers, and the uneven deposition of animal excreta (McCarl and Schneider, 2000; Mosier *et al.*, 2001; Bolan *et al.*, 2004). Koops *et al.* (1997) established that in grazed grasslands there is a rapid and intense nutrient cycling. This would of course have a direct relation with

the amount of grass produced, the stocking rates and in the intensity of grazing. For these reasons a good management of the system should consider N inputs and N outputs that are generally attributed to the volatilization of  $\text{NH}_3$  from leaves of rich plants, soil denitrification-nitrification and soil leaching (Raun and Jonson, 1999; Glass, 2003), therefore, a suitable management to reduce N losses should include all of the components already mentioned.

The fertilizer is often applied in a very inefficient way to the production systems, where the greatest part never reaches the cultivated plants and for such reason instead of fulfilling a role in the development of these, they cause problems in the fields and streams by leaching or become gas, as the  $\text{N}_2\text{O}$  escape to the atmosphere or its volatilization like  $\text{NH}_3$  (Nierenberg, 2001). The main concern with N fertilization is related to the minimization of the losses, either by  $\text{NH}_3$  volatilization resulting from the decomposition of N fertilizers or by leaching of  $\text{NO}_3^-$  which are weakly retained by the solid phase of the soil (Goedert *et al.*, 1997).

Smits *et al.* (2003) found that grazing was influencing  $\text{NH}_3$  volatilization in a range of 1.4-15%. In a field study with cows, substantial differences were obtained in emissions from grazing. This difference was caused by the numbers of grazing-hours during summer (ranged between 3.5 and 20 h per day), and partly these differences are related with the dietary concentration. For example, in The Netherlands, the number of grazing-hours is currently tending to decline, as mineral inputs and outputs can be managed more accurately when cows are fed indoors. As a result, their excreta are spread better on crops, and also in a better balance and with better timing in regard to their uptake by crops.

The grazing livestock causes an immediate increase in  $\text{N}_2\text{O}$  by the deposition of urine patches on the soil (Carter, 2007). In this case Passianoto *et al.* (2003) demonstrated in pastures, under different managements, that the tillage systems in the grassland affect the production of  $\text{N}_2\text{O}$ . Tillage treatments ( $0.94 \text{ kg N ha}^{-1}$ ) were compared with no-tillage ( $0.64 \text{ kg N ha}^{-1}$ ). The estimations of the emissions from the control pasture were of  $0.07 \text{ kg N}_2\text{O ha}^{-1}$ . These results could be explained by Aulakh *et al.* (1984) and in a study achieved in Canada and the United Kingdom (Ball *et al.*, 1999a). These studies showed an increased soil water content, bulk density, and larger soil aggregates, which led to increases in anaerobic conditions.

Saggar *et al.* (2004a) indicated that incorporating slurry into soil may reduce  $\text{NH}_3$  emissions, but such slurry applications may enhance  $\text{N}_2\text{O}$  emissions. This would imply an increase in the emission of a conservatory gas and a reduction in  $\text{NH}_3$  volatilization. The situation is explained by Brink *et al.* (2001) where they indicated that  $\text{NH}_3$  abatement may have an adverse effect on  $\text{N}_2\text{O}$  emissions, while abatement of  $\text{N}_2\text{O}$  results in a net decrease in  $\text{NH}_3$  volatilization.

In order to reduce gas emissions, animal waste is applied to the grassland. Good practices of waste-management and continuous measurements that reduce surplus substrates during storage, such as composting and digestion, have to be implemented and so these tend to diminish the emissions from the soil (Saggar *et al.*, 2004a). Wu *et al.* (2003) indicated that an inadequately handled animal manure is a significant threat to the quality of the air and to nearby water of the soil, the storage and disposal area. That is why these authors recommend that management's practices in these cases should determine the source of N and the concentration before any application to the soil. Among the strategies raised to reduce the emissions is the reduction of N gases from grazed pastures that should focus on reducing the gases emitted from animal

excretes. Another mitigation option would be to manipulate the N concentration in the forage in order to reduce the N excreted. A lower N content of pasture would reduce N excretion by animals and  $\text{NH}_3$  volatilization. Another method is to combine the urease enzyme and the nitrification inhibitors to reduce the  $\text{N}_2\text{O}$  emissions and result in a small decrease in  $\text{NH}_3$  emissions. Restrictions on grazing can reduce emissions from dairy-farming systems (Oenema *et al.*, 1998; Saggar *et al.*, 2004a; Velthof and Oenema, 1997). The form of application of animal waste and the time of application is also worth improving.

### ***Rotational grazing***

In rotational grazing, N volatilization is very important, and also is the emission of other gases from the soil. Many dairy production systems rely strongly on a grassland feeding resource that is exploited through rotational grazing (moving animals from one pasture to another) and complemented by conserved food (maize, silage and hay) and concentrates, especially in winter when the herbage mass is insufficient (Cros *et al.*, 2001). Rotational grazing management problems may change from one year to another because the stock of maize available may differ and the size and characteristics of the herd may also vary.

Losses of  $\text{NH}_3$  were greater from pastures during and immediately after grazing, and the highest rates of loss are associated with high stock densities under a rotational grazing system (Saggar *et al.*, 2004a; Ryden and McNeill; 1984). This indicates that the density of animals is an influential factor in the produced volatilization, making emphasis on the moment at which the pasturing ends (Saggar *et al.*, 2004a). The bigger losses from the high-N ryegrass pastures were attributed to the higher number of stock used, and also to the greater proportion of ingested N being returned in the form of urine. There is a direct relation in the concentration of ingested N, the amount of

animals by area, the amount of urine and dung deposited and the amount of N that will leave to the atmosphere.

### Livestock grazing effects on pasture nutrient cyclins

The grassland possess a great potentiality for gas emissions to the atmosphere (Jarvis, 1993) and when having added high doses of N to the soils, the possibilities of leaching, denitrification and volatilization increases (Jarvis *et al.*, 1991). In this sense, Denmead *et al.* (2004) indicated that the emission of NH<sub>3</sub> in grassland under grazing changes with the season, thinking that under the conditions of the study, NH<sub>3</sub> emissions in summer were of 14.8 kg ha<sup>-1</sup> and in autumn 2.0 kg ha<sup>-1</sup>.

In the livestock system, animal excretes is a source of nutrient return for the soil (Beetz, 2001; McGechan and Topp, 2004; Beetz and Rinehart, 2006), even though in small and focalized areas. The areas of excretion inside the grassland represent between 30 to 40% (Haynes and Williams, 1992) or 15-31% (Kear and Watkinson, 2003). The latter percentages are very low in comparison to 60% for dung and 5% for urine reported by Saggar *et al.* (1988). Therefore, the effect of the urine and dung of cattle is very changeable and the return in the grassland will depend on many factors, including the concentration of entry and exit of the productive areas.

The management practices of the grassland, especially the fertilization affect the cycle of the nutrients, and Burke *et al.* (2002) indicated that the cropping practices, much more than grazing, result in substantial alterations of N budgets, through fertilization, enhanced N cycling, enhanced N trace gas losses, and increasing NO<sub>3</sub> leaching. A mechanism to reduce N losses is the application of the concept of carrying capacity that is defined as the number of organisms that can support an ecosystem, and in our case, the number of cattle heads

that the pasture can support. The carrying capacity is determined by a variety of factors, such as food supply, nesting sites, water supplies, climate conditions and waste assimilation (Chiras *et al.*, 2002). Beetz (2001, 2002) and Beetz and Rinehart (2006) established that animals use very few of the nutrients from the plants they eat; most minerals are returned in animal wastes and can be considered part of a natural cycling of nutrients. Nevertheless, the volatilization refers to nutrients carried off by air and can be a problem in pastures if supplemental fertilizer applications are applied at the wrong time or under the wrong conditions. Grazing animals have a major role in the cycling of nutrients and are responsible for rate increments at which nutrients are cycled (Willian and Haynes, 1990).

The grazing directly affects the N cycle by removing plant biomass and by returning in a significant proportion (50 to 75%) as urine and dung. Therefore, increasing the rate of N in the cycle may result in an increment of N losses from ecosystems (Burke *et al.*, 2002). These researchers reported that there is loss of N by excess of stocking rates by different pathways as NH<sub>3</sub> volatilization, or as leached NO<sub>3</sub> (25-44 kg N ha<sup>-1</sup> into surface ground water). The reason for the alteration of the nutrients cycle by effect of the stocking rates is demonstrated in the grassland (Burke *et al.*, 2002).

Practices that favour effective N use and cycling in pastures include (Bellows., 2001): 1). maintaining stable or increasing percentages of legumes by not overgrazing pastures and by minimizing N applications, especially in spring., 2) protecting microbial communities involved in organic matter mineralization by minimizing practices that promote soil compaction and soil disturbance, such as grazing of wet soils and tillage, 3) incorporating manure and N fertilizers into the soil, and never applying these materials to saturated, snow-covered, or frozen soils, 4) avoiding pasture burning.

If burning is required, it should be done very infrequently and by using a slow fire under controlled conditions, 5) applying fertilizers and manure according to a comprehensive nutrient management plan.

### **Losses of nutrients caused directly by animal grazing**

Animal grazing can result in nutrient losses from the grassland and these losses happen due to the elimination of N pathway excretions, conversion to products like milk, meat, wool, fiber, leather or utilization in its metabolic processes (Willian and Haynes, 1990), since nutrients are lost by dung and urine, they might not be recycled appropriately, which is the case of N drainage, depositions during dairy and deposition outside the grassland fields (e.g. B-roads, freeways).

The experience of Di *et al.* (2002) is that in the case of the urine, 50% of the losses happen during the two first days after the application of the urine, or after that the dejection occurs. The animal grazing will have negative or positive effects in the cycle of the nutrients, depending on the managing that is applied in the production system. The benefits would be; a major utilization of the pasture, rapid decomposition in the animal and incorporation in the cycle of the nutrients. The negative effects might be; losses of nutrients, concentration of nutrients in small volumes of the soil and difficulty for the plants to recover and uptake. The animal grazing has a dominant effect on the movement of nutrients in the soil/plant/animal of the system and the fertility of the soil under pasture. The major emphasis of the animal grazing is to influence the fertility of the soil by the contribution of nutrients (Haynes and Williams, 1993). Hydrolisis of the urea produces a high environmental pH and this produces NH<sub>3</sub> volatilization. In the case of urine, hydrolysis is usually completed after

three days of the deposition when the greatest emission is produced (Ryden, 1984; Lantinga *et al.*, 1987).

The amount of N volatilized as NH<sub>3</sub> is considerably different with the period of grazing since it will depend on the amount of N excreted via urine and dung per period and also on the dose of the applied fertilizer. Generally, the losses are high only days after the grazing and decrease to a minimal amount after ten days of the grazing. Nevertheless, Harper *et al.* (1996) indicated that the volatilization of NH<sub>3</sub> in spring and summer is similar. However, in other researches it has been proven that the season influences the volatilization of NH<sub>3</sub> in grassland under grazing, with emissions much higher in summer than in winter (when the lowest emissions occur) by effect of the temperature and N concentration in urine (Misselbrook *et al.*, 1998a; Misselbrook *et al.*, 1998b; Smith *et al.*, 2003; Pinder *et al.*, 2004). Besides, it has also been established that the extension of the season of grazing also affects the emission of this gas (Webb *et al.*, 2005a).

### **CONCLUSIONS**

The grazing livestock provokes negative impact on the environment; especially gases emission like NH<sub>3</sub> and N<sub>2</sub>O, mainly due to an excessive use of N fertilizers and the increase in the stocking rates. These losses represented between 20-30% of the N inputs from animal-plant-soil system. In the case of NH<sub>3</sub>, losses can represent between 8-9% of the inputs in temperate grassland and between 12-21% in tropical regions. The emissions of NH<sub>3</sub> shown are very variable among the countries with similar environmental conditions, type of grassland and management. The average in countries like New Zealand, England and Australia were between 40-50 kg ha<sup>-1</sup> yr<sup>-1</sup>. These emissions are high from an economic and environmental point of view.

The N<sub>2</sub>O is less than NH<sub>3</sub> emissions at global scale. The lower value level was about 10 kg ha<sup>-1</sup> yr<sup>-1</sup> and it represents less than 3% of the N inputs in the system. The emissions of this gas in the livestock systems depend on environmental factors and managements conditions. Different scenario during the season will produce changes in the emission, for example, over grazing, variation of soil moisture, grassland management and stocking rate.

This review showed the existence of several techniques for measuring of NH<sub>3</sub> and N<sub>2</sub>O losses: chamber and micrometeorological techniques are the most often used. Indirect open measurement technique, such as <sup>15</sup>N and N balance are also popular. The application of these methodologies varies in their accurate level, costs and instrumentation, because the factors that influence the emissions. Advantages and disadvantages of these techniques were discussed early in the review.

The general strategies to reduce the emission of NH<sub>3</sub> and NO<sub>2</sub> gases are: regulation of the stocking rates, the amount of N fertilization, timing and the type of fertilizer used. Other mechanisms are pH regulation of the animal urine with adipic acid, regulation of the animal's diet and N concentration in the forage.

Finally, fertilization management and strategies for controlling the NH<sub>3</sub> and N<sub>2</sub>O emissions are being used, e.g. the uses of inhibitors (urease inhibitor, nitrification inhibitors) as much as for the nitrification and denitrification with the aim of inhibit some biological processes that induce excessive losses. In the case of denitrification, inhibitors reduce until upto 80% of N<sub>2</sub>O losses. However, these inhibitors have a negative effect in the microbial communities of the soil and for this reason they are being currently investigated. We need more results to ensure the use and increase all our technology in order to minimize the impacts on the environment.

## ACKNOWLEDGEMENTS

The carrying out of this investigation was possible thanks to the Dominican Republic Institute of Agricultural and Forest Research (IDIAF) and Universidad de La Frontera of Chile for its financial support.

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