CHAPTER FOUR

Livestock Production and Its Impact on Nutrient Pollution and Greenhouse Gas Emissions

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Abstract

The livestock sector provides more than one-third of human protein needs and is a major provider of livelihood in almost all developing countries. While providing such immense benefits to the population, poor livestock management can potentially provide harmful environmental impacts at local, regional, and national levels which have not been adequately addressed in many countries with emerging economies. Twenty-six percent of global land area is used for livestock production and forest lands are

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continuously being lost to such activities. Land degradation through soil erosion and nutrient depletion is very common across pastures and rangelands. The intensification of livestock production led to large surpluses of on-farm nitrogen and phosphorus inputs that can potentially contribute to nonpoint source pollution of water resources in many parts of the world. The sector is one of the largest sources of greenhouse gases (GHGs) contributing around 14.5% of all human-induced GHG emissions, a major driver of use and pollution of freshwater (accounting 10% anthropogenic water use) and a contributor to the loss of biodiversity. About 60% of global biomass harvested annually to support all human activity is consumed by livestock industry, undermining the sustainability of allocating such large resource to the industry. Despite the negative impacts of livestock production, opportunities exist to balance the competing demands of livestock production and the environment. These include (1) improved technologies and practices that increase livestock productivity with optimal use of land and water, (2) reorienting grazing systems to provide environmental services for water, biodiversity, carbon sequestration, and resource conservation, (3) reducing GHG emission from livestock production, and (4) an effective management strategy for efficient and sustainable use of manure in livestock production. Further research, appropriate policy development, and institutional support are important to ensure the competitiveness of the industry. Integration of crops with livestock production provides opportunities for increasing resource use efficiencies and reducing environmental pollution, making the system resilient to impacts of climate change, reducing GHG emissions from the system, enhancing soil quality and fertility, and improving water quantity and quality. Appropriate techniques for assessing and monitoring impacts of livestock production are necessary for developing strategies and making the system profitable, sustainable, and resilient. Isotopic and nuclear techniques play an important role in such assessment and monitoring.

1. INTRODUCTION

The global livestock production is increasing rapidly as the demand for livestock products for human consumption increased. In the past few decades, this growth was achieved through sufficient land availability and concentrated animal operating systems (FAO, 2006a,b). This increase in production is expected to continue into the future as real income grows in developing economies. Currently the sector accounts for 1.4% of world's gross domestic product (GDP). It is a major contributor to human nutrition (protein) and health and provides a buffer against grain shortage assuring food security to human population (Smith et al., 2012).

Livestock production is a major provider of livelihoods for larger part of the world's poor, particularly in dry areas, and plays a crucial role in rural economies of most developing countries (Godfray et al., 2010;

McDermott et al., 2010). It provides food for 1 billion of the world's poor and contributed an average of more than 40% of global agricultural output (Steinfeld et al., 2006). With more than US $460 billion GDP in major agricultural economies (Table 1), more than 1500 million ha land area is currently used for livestock production through pasture management.

In the next 10–15 years the livestock sector is expected to provide 50% of agricultural output in value terms. While global demand for food is expected to grow by 50% over the next 20 years to feed the growing human population, the demand for livestock products is expected to double during the same period (Thornton, 2010). Since the 1980s the demand for livestock products is stronger than for most other food items. Global meat production is projected to increase from 229 million tons in 1999/2001 to 465 million tons in 2050, and that of milk to increase from 580 to 1043 million tons (FAO, 2006a,b). Throughout the developing world, the integration of livestock with crops in the same farm (mixed crop-livestock systems) represents the backbone of family farming and provided 50% of the world's meat, over 90% of its milk, and 50% of its cereals (Thornton and Herrero, 2010). For many poor farmers in developing countries, livestock manure is also a source of renewable energy and is an essential source of organic fertilizer for their crops.

Meeting the increased demand for livestock products may put substantial pressure on land and water resource use and biodiversity conservation.

Table 1 Value of Livestock Production in Selected Countries (FAO, 2005)

Country	Agricultural Land (Million ha)	Pasture Land (Million ha)	Livestock GDP (Billion US $)	% of Agricultural GDP
Argentina	129	100	5	37
Australia	455	408	61	34
Brazil	264	197	33	48
China	550	440	61	34
European Union	141	56	120	40
India	170	12	20	20
New Zealand	18	14	7	87
South Africa	100	84	3	46
United States	412	233	150	39

Together with climate change and climate variability, these demands for resource uses add up to a formidable set of development challenges for both developed and developing countries. All together about 70% of agricultural land is currently used for livestock production either directly (grazing) or indirectly (concentrated livestock operations). Conversion of forests to cattle ranches is the biggest cause of deforestation in the Brazilian Amazon (Fearnside, 2008). Seventy percent of deforestation in the Brazilian Amazon is used for medium- to large-sized cattle ranches (Fearnside, 2005), and around 80% of the total deforested land is used for cattle grazing putting considerable stress on forest ecosystems (Greenpeace, 2009). Studies have shown that in countries such as the United States livestock production was estimated to account for 55% of soil and sediment erosion, and more than 30% of the total nitrogen (N) and phosphorus (P) loading to national drinking water resources (WRI, 2009). Livestock operation is considered as a primary accelerator of global nutrient cycling (Bouwman et al., 2009). International assessments showed that between 2000 and 2050, global livestock production will increase by 115% leading to 23% increase in global N and 54% increase in P surpluses (Bouwman et al., 2013). Most of the surplus N is then lost to the environment as volatilization, denitrification, leaching to groundwater, and runoff to surface waters, while the surplus P is lost by runoff to waterways and causes eutrophication leading to water quality issues. In addition to land and water quality degradation, livestock production emits greenhouse gases (GHGs) approximately 7.1 million tons of CO_2 equivalent per annum (consisting of 44% methane (CH_4), 29% nitrous dioxide (N_2O), and 27% carbon dioxide (CO_2); http://www.fao.org/news/story/en/item/197623/icode/), which represents 14.5% of all human-induced emission (Gerber et al., 2013). As resources required for sustaining the growth of livestock production are strained, future increase in livestock products must be accommodated within the existing resources including land, water, and nutrient (FAO, 2011). Improving resource use efficiency and reducing environmental foot prints (including impacts on climate) of livestock production are paramount for the sustainability of the sector.

This chapter reviews the environmental impacts of livestock production, in particular the soil quality, GHG emission, and nutrient losses as influenced by grazing and manure management and opportunities to make livestock production sustainable for agricultural intensification, resource use efficiency, and environmental protection including climate change.

2. LIVESTOCK FOR ECONOMIC DEVELOPMENT AND CONSUMER DEMAND

Global livestock production is changing rapidly in response to: (1) population growth which is projected to exceed 9 billion by 2050, (2) rapid urbanization, and (3) growing incomes of people and their nutritional food requirement. It is expected that by 2030 about 5 billion people live in urban areas with unprecedented urban growth in Africa and Asia (UNFPA, 2008), leading to an increased demand for livestock products (Delgado, 2003). With significant contributions to global agricultural production, food supply, nutrition, rural employment, and soil fertility, livestock plays an important role providing income, poverty reduction, and livelihoods, among the rural population in developing countries (Moyo and Swanepoel, 2010; Perry and Sones, 2007; Randolph et al., 2007). Between 1950 and 2000 per capita income of rural communities grew annually at a rate of 2.1% globally. The FAO and World Bank reports showed that the expenditure on livestock products grows with income (Steinfeld et al., 2006). About 600 million poor people in Africa, Asia, and Latin America and Caribbean (LAC) depend on livestock for food and milk products, more than 1 billion people rely on livestock-based livelihoods, and more than 1.3 billion people are employed in livestock industry making it socially and politically significant in developing countries (Kristjanson et al., 2007). Between 1990 and 2002 global per capita meat and milk consumption increased by 55% and 20%, respectively, in developing countries and is projected to increase by 30% and 45%, respectively, by 2030 (FAO, 2006a,b, 2012). Rosegrant and Thornton (2008) reported that between 2000 and 2050 per capita consumption of meat is projected to increase in all regions of the world and by up to 100% in Sub-Saharan Africa making livestock industry a significant contributor to the global food and nutrition security.

While the per capita meat and milk consumption in the developed countries is much higher than that in the developing countries (78 vs 28 kg/year and 202 vs 46 kg/year for meat and milk, respectively) during 1990 and 2002 (FAO, 2006a,b, 2012), the projected meat and milk consumption in developing countries will exceed (by 64% and 18% for meat and milk, respectively) the developed countries by 2015 (Fig. 1). Such an increased demand for livestock products comes with enormous challenges for the sustainability of the industry including the protection of land and water resources, managing the manure, and reducing GHG emissions from the

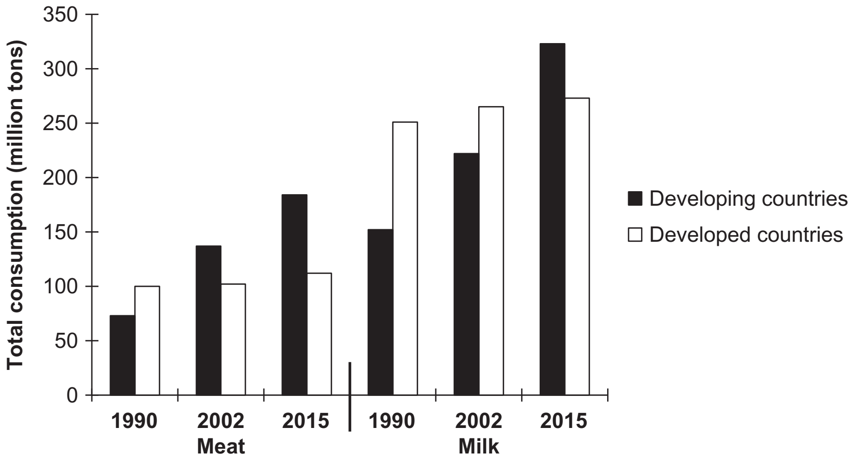

Fig. 1 Change in livestock product consumption between 1990 and 2015 in developing and developed countries (Thornton and Herrero, 2010).

production system. With the existence of heterogeneity and complexities in the system, converting the extensive livestock system to intensive and sustainable production system is important to reduce negative impacts of livestock production on the environment (Havlík et al., 2014) including GHG emissions.

In marginal semiarid and arid rangelands, the impacts of climate change (increased temperature and unpredicted rainfall) lead to frequent droughts and reduce the livestock productivity. This is notably true for southern Africa and Central Asia and causes soil water deficits, leading to overall productivity decline. In many other parts of semiarid region with no grazing (cut, carry, and feed systems), ground water is mainly extracted for fodder production and is driven by productivity gains and economic benefits (FAO, 2011). However, the sustainability of using such large volumes of groundwater for livestock production has been questioned.

3. LIVESTOCK PRODUCTION AND LAND USE CHANGE

Livestock production is the largest land user on earth (directly or indirectly) and uses about 30% of the earth's entire land surface, and subsistence farming is part of this land use in many developing countries across the globe, and in many part of the world, it is one of the main contributors to deforestation (Pan et al., 2007). Feed requirement links livestock production with land use which depends upon the type of feed used by livestock in a particular region (Herrero et al., 2013). In some countries, up to 85% of agricultural land is used for livestock production and contributes to a significant proportion of agricultural GDP (Table 1). In the last 200 years, grazing land expanded sixfold including areas in North America, South America, and Australia, where little or no livestock grazing occurred previously (Asner et al., 2004). Currently grazing land occupies about 26% of the global land area, and about 33% of all crop land is used for animal feed production (Steinfeld et al., 2006). Such a large allocation of land area for livestock production needs to be balanced with environmental services, biodiversity conservation, and socioeconomic developments.

The poor feed quality in developing countries such as those in Sub-Saharan Africa means that livestock is mainly fed with nutrient deficient grasses and crop residues. However, in the developed countries (e.g., the United States and Europe), livestock production is intensive and nutrient-rich feedstock is used. As a result the amount of feeds consumed by livestock in resource poor countries can be 10 times more than that consumed by livestock in developed countries to produce the same amount of protein. This means more land area is required for producing same amount of livestock in developing countries compared to developed countries. Because of this, more land needs to be cleared from forest contributing to deforestation. Additionally, more water resources are required to meet such a demand. Normally 1 ha of land can be used to grow crops (rice, potato, etc.) which can be used to feed up between 19 and 22 people annually. However, if the same area is used for meat production, only two people can be fed annually. Given that per capita land availability is below 0.30 ha globally, it is very difficult to sustain livestock production systems, particularly in the developing world where a majority of marginal farmers live. Tropical deforestation progresses at a high rate with serious consequences for the environment. Livestock's role in deforestation is of particular importance in Latin America where the largest net losses of forests and carbon from landscape occurred (Barona et al., 2010; Fearnside, 2005, 2008). In tropical Latin America and sub-Saharan Africa, there is a rapid expansion of pastures for ranching, with 0.3–0.4% of forest lost to pasture annually (FAO, 2009), and such an expansion of pasture and arable land for feed production occurred largely at the expense of forest area (Table 2).

Between 1961 and 2001, global arable and pasture lands increased (0.1–0.3% and 0.2–0.3% per annum for arable and pasture lands, respectively) at the expense of forest lands (0–0.1% per annum). In Central America, forest area has been reduced by almost 40% over the past four decades,

Table 2 Annual Land Use Change (%) in Latin America and the Sub-Saharan Africa From 1961 to 2001 (FAO, 2005, 2006a,b)

	Latin America			Sub-Saharan Africa		
Year	**Arable Land**	**Pasture**	**Forest**	**Arable Land**	**Pasture**	**Forest**
1961–1991	1.1	0.6	−0.1	0.6	0.0	−0.1
1991–2001	0.9	0.3	−0.3	0.9	−0.1	−0.5
Share of total land (%)	7.4	30.5	47.0	6.7	34.7	27.0

(−) Sign indicates reduction in land use.

with pasture and cattle population increased rapidly over the same period. During the period between 2004 and 2005 an estimated 1.2 million ha of rainforest was cut down in Latin America for soybean expansion, a majority of which is used for animal feed production (Wassenaar et al., 2007). According to Brazilian National Institute of Space Science, about 18.9 million ha were deforested during 2000–2004 and up to 66% were for pasture (Barona et al., 2010). Over the period between 2000 and 2010 an average of 2.4 million ha of pasture land expanded on an annual basis at the expense of forest land in Latin America. The Intergovernmental Panel on Climate Change estimated that approximately 1.7 billion tons of CO_2 per year is released to the atmosphere as a result of deforestation (IPCC, 2001), and more than 65% of this deforestation is used for animal production through either grazing or growing feed for animals. During 2010 and 2011, an estimated 6200 km^2 of Amazonian forest has been cleared for agriculture, ranching, and soybean for livestock feed (INPE, 2010; Nepstad et al., 2006). The period 2002–2004 saw historically high rates of deforestation in Amazon primarily driven by beef and soybean production (Nepstad et al., 2006). Similarly, in Indonesia about 1 million ha of tropical forest has been removed through deforestation mainly for agriculture (Hansen et al., 2009). Throughout the tropical areas, about 10 million ha of forest has been cleared in 2010 (Houghton et al., 2012) for agriculture and grazing. The social, economic, and environmental costs of deforestation are enormous and are responsible for increased carbon dioxide emission to the atmosphere from land. In addition, the loss of biodiversity, soil erosion, and water quality degradation lead to less environmental well-being of the landscape and the population. It is clear that sustainable livestock production with reduced land clearance in tropical region is important for enhancing resource use efficiency, reducing land degradation, and improving sustainability of livestock production.

4. ENVIRONMENTAL IMPACTS

While providing significant social benefits in the form of improved nutrition and poverty alleviation, livestock production has often been subjected to substantial public scrutiny as a result of its impact on the environmental quality. In the process of providing social benefits, livestock production uses large quantities of water and fertilizer and also emits significant amounts of GHGs (Herrero et al., 2009). The negative impacts of livestock production increasingly become serious at local, regional, national, and global levels from land degradation and water pollution to the loss of biodiversity and climate change (Herrero and Thornton, 2013). The sector contributes approximately 14.5% (7.1 billion tons CO_2 equivalent) of the global GHG emissions and accounts for 9% of anthropogenic CO_2 emissions (1.92 billion tons CO_2 equivalent), most of it is due to the decomposition and mineralization of soil organic matter (SOM) resulting from land clearance and the expansion of pastures and arable land for feed crops (Gerber et al., 2013). It generates even bigger shares of emissions of methane (CH_4) and nitrous oxide (N_2O) with the greater potential to warm the atmosphere. As much as 35% of anthropogenic CH_4, mostly from enteric fermentation by ruminant, and 65% of N_2O, mostly from manure and nitrogen fertilizer management, emitted to the atmosphere (http://www.fao.org/agriculture/lead/themes0/climate/en/). Major environmental impacts of livestock production arise from the following: (1) manure management, (2) deforestation, (3) unsustainable use of freshwater, (4) nutrient pollution, and (5) GHG emissions. These are briefly discussed in the subsequent sections.

4.1 Manure Management

Livestock manure is a valuable source of nutrient for improving soil fertility and quality through enhancing SOM. The global manure production through the concentrated animal feeding operations has increased tremendously in recent years. Based upon animal density and international data, estimates showed that approximately 128 million tons of N and 24 million tons of P equivalent of manure produced annually (Potter et al., 2010). In the United States alone, approximately 24.4 million tons of manure per year came from beef cattle, 19 million tons of manure per year from dairy farming, 12.7 million tons per year of litter and manure from poultry, and 14.5 million tons per year from swine are produced. It has been estimated that about 7.5 million tons of N and 2.3 million tons of P are generated in the United States annually by the livestock industry compared to 9 million tons N and 1.6 million tons P applied to agricultural land in the form of commercial fertilizers (http://rcrec-ona.ifas.ufl.edu/in-focus/IF7-21-06.shtml). In confined livestock production systems, the feed is almost entirely brought from the field (sometimes even transported from far). However, in the production process, animal products (for example, meat and milk) are removed, while the manure is left behind resulting in the disconnection between manure and the production area (the field). This leads to transportation costs and the associated GHG emissions if they are applied back to the field. In some situation, the manure is disposed without any environmental consideration to reduce costs.

The enormous increase in manure production from livestock enterprises has generated environmental concerns due to limited land area for efficient manure application and spreading. While it is possible that the N and P in manure react slowly compared to mineral fertilizer N and P, they may be over applied to some areas leading to nutrient enrichment. This will increase nutrient losses and nitrous oxide (N_2O) and methane (CH_4) emission from agricultural lands. Manure management contributes to approximately 7–8% of agricultural GHG emission mainly as N_2O and CH_4 (IPCC, 2014). Water quality is increasingly become an important issue when manure application exceeds vegetation/crop requirements (Ribaudo et al., 2007). This mismanagement of manure often leads to direct discharge of liquid manure to surface waters through runoff and causes eutrophication which is characterized by high concentrations of N and P creating ecological imbalance in the water system. Such runoff could happen immediately after application depending on weather conditions (Smith et al., 2007a,b). Judicious and sustainable application of animal manure to land will contribute plant nutrients to crops and reduce the need for mineral fertilizers. However, in many countries, sustainable and environmentally sound manure management (production, storage, and application) practices have been very slow due to the fact that manure has been considered as a waste rather than a resource. There is every opportunity to reduce pollution risks (N and P input to surface and ground waters) and soil quality improvements through best practice livestock and manure management. Understanding the dynamics of nutrient applied through manure under a changing environmental conditions helps to develop management practices which will reduce environmental pollution risks of manure in livestock production systems.

4.2 Water Use in Livestock Production

Worldwide livestock production is a leading consumer of water and is divided into two categories of consumption, namely, (1) upstream that includes water used for forage and feed production and on-farm drinking water under grazing, and (2) the downstream includes water used for livestock processing such as making meat and milk products (Doreau et al., 2012; Ran, 2010). The sector accounts for approximately 10% of the global anthropogenic water use. This water demand for livestock production is influenced by several factors that include type of animal, its activity, the feed, and the quality of water available for the livestock (Lardy et al., 2008). Grain-fed beef production takes about 50 times the water required to produce the grain. Raising broiler chickens takes 3500 L of water to make a kilogram of meat. In comparison, soybean production uses 1500 L for kilogram of food produced and rice about 1700 L of water per kilogram of rice (http://www.news.cornell.edu/). Estimated daily water consumption for a range of livestock production is provided in Table 3. Water shortages already are severe in the arid and semiarid regions, including Western and Southern United States, and the situation is quickly becoming worse because of a rapidly growing population that requires more water for all of its needs, especially agriculture. Unsustainable use of freshwater for feed production, animal care, and slaughterhouses contributes to water scarcity and is depleting water resources in many agricultural landscapes (Burkholder et al., 2007; Walker et al., 2005).

Table 3 Typical Daily Livestock Water Consumption (Brown, 2006; NSWDPI, 2007)

Type of Animal	Water Consumption (L/Day/Head)
Sheep	
Weaner	2–4
Ewes with lamb	4–10
Cattle	
Lactating cow	40–100
Young stock	25–50
Swine	
Weaner	7–23
Feeder	23–113

Most critical for livestock production in arid/semiarid areas is the availability of water. Livestock feed production accounts for about 45% of global agricultural water use (Zimmer and Renault, 2003). However, water use efficiency is low in grasslands and noncultivated fodder lands used for grazing. In a majority of livestock production systems in these areas, water management has been practiced without consideration to the interaction between livestock and water. While regional variation exists, the average water use efficiency in the production of barley, maize, wheat, and soybean for feed is only 10%, with Sub-Saharan Africa at 1% and Western Europe ranging from 25% to 29% (FAO, 2006a,b). Water used to produce fodder (indirect) is generally at least 50 times greater than water directly used by livestock. In general the water footprint of livestock products is larger than that of crop with equivalent nutritional value (Mekonnen and Hoekstra, 2012). For example, the average for beef is 20 times larger than for cereals and starchy roots. The sustainability of such practices is questionable as soon as groundwater is used to irrigate fodder crops (Steinfeld et al., 2006). In Australia, about 23% of dairy farms are irrigated and about 52% dairy farmers use supplemental irrigation with natural rainfall which will put enormous pressure on agricultural water in a country which has limited water availability (Dairy Australia, 2006; Nash and Barlow, 2009).

Practices and technologies that improve the sustainability of water use in livestock production are important. These include (1) livestock feed sourcing, (2) grazing and watering strategies, and (3) integrating livestock with cropping systems that uses crop by-products (crop residues) by livestock as a major water saving practice. Reducing irrigation for feed production during certain period of the growing season also improves water sustainability and productivity in livestock production. With more than 40% contribution to gross agricultural productivity, any improvement in water use efficiency in livestock production will improve overall agricultural water productivity and environmental water footprints.

4.3 Livestock and Soil Physical and Chemical Characteristics

The binding of soil particles with organic matter into aggregate is the major physical characteristic of soils that influence its function and suitability as a medium for plant growth and regulate the movement of air, water and nutrient in the soil, water retention, and physical environment for active microorganisms and plant roots (Cuttle, 2009). Global climate change alters rainfall regimes with the possibility of increased occurrence of droughts and higher frequencies of extreme rainfall events during the crop growing season. The spatial and temporal variability of rainfall increased the frequency of wetting and drying cycles which is likely to affect soil water content (Harper et al., 2005). The presence of grazing animal further accelerates the wetting and drying cycles (Cuttle, 2009). The resultant drying and wetting of soil is expected to modify soil structure through the physical processes of shrink–swell (Carter and Stewart, 1996) and affect water infiltration and GHG emissions under grazing systems.

Soils with large clay contents (>60%) shrink as they dry and swell when they become wet again, forming large cracks and fissures. Crack formation results in rapid and direct movement of water and solutes (nutrients, metals, and dissolved organic matter) from surface soil to the unsaturated zone through bypass or preferential flow leading to possible losses of nutrients and water pollution from soil root zone which is generally available to plants (Nguyen et al., 1998; Rounsevell et al., 1999). Dry soil conditions for longer periods of the year would also result in intensification of wet upland grazing areas due to livestock concentration (Rounsevell et al., 1996). When the soil becomes wet, it is subjected to structural alteration under the influence of grazing animals, and this alteration increases with increasing soil water content (Mulholland and Fullen, 1991). This change in soil water content associated with soil wetting and drying cycles and livestock grazing affects a number of soil processes including hydrology and vegetation growth. Soil structural alterations associated with grazing animals including compaction, pugging, and poaching (Bilotta et al., 2007).

Compaction reduces soil pore space and permanently removes air and water from the soil leading to mechanical disruption of soil aggregate, reducing aggregate stability, increasing the bulk density and penetration resistance of the soil, and creating anaerobic conditions in the soil (Abdalla et al., 2009; Donkor et al., 2002; Mwendera and Mohamed Saleem, 1997). As the drainage is impeded due to poaching, the soil becomes prone to surface runoff and erosion leading to sedimentation and water quality problems downstream (Mulholland and Fullen, 1991).

Soil infiltration can be reduced by up to 80% under the influence of grazing. For example, Trimble and Mendel (1995) reviewed the impact of livestock on soil infiltration and showed that it can be decreased from approximately 50 mm/h on lightly grazed to 25 mm/h on heavily grazed lands. Similarly, Heathwaite et al. (1990) showed that infiltration capacity was reduced by 80% and surface runoff increased by 12 times on heavily grazed compared to ungrazed pastures. In Alberta, Canada, the bulk density and penetration power of soil were significantly greater by 15% and 17% in short duration grazing with 4.16 animal unit months per hectare compared to continuous grazing with 2.08 animal unit months per hectare, suggesting that the duration of grazing influences the characteristics of the soil (Donkor et al., 2002). Studies carried out in tallgrass prairie pastures showed that for ungrazed pasture the largest infiltration was 29.5 cm/day for a clay loam soil and 27.5 for a silt loam soil, while the grazing treatments had lower infiltration irrespective of soil types (Daniel et al., 2002). In northern China, soil characterization under grazed and ungrazed systems has shown that the bulk density significantly increased under grazing (Zhou et al., 2010).

Changes in soil physical characteristics can influence the processes affecting soil processes and nutrient transformations by altering soil moisture, soil redox potential, and plant uptake processes (Bilotta et al., 2007; Di et al., 2001). For example, in both temperate and tropical regions the presence of grazing animals significantly increased soil erosion under pastures (Hamza and Anderson, 2005). The National Land and Water Resources Audit in Australia has found that soil erosion from native pastures (with grazing) in the northern region of Australia accounts for 76% of continent's total soil erosion, and across these grazing lands the rate of soil loss is several times greater than the soil formation (NLWRA, 2001). As many as 100 million ha of rangeland is considered highly erodible in the United States (USDA-NRCS, 1992). Improved technologies and grazing management practices can help reduce impacts of soil physical and chemical characteristics on soil erosion. For example, in the United States, soil erosion losses from crop lands have been reduced from 1.2 million tons per year in 1992 to 960 million tons per year in 2007 (NRCS, 2010).

Within the context of agricultural productivity the most obvious chemical characteristics influenced by grazing are nutrients (nitrogen and phosphorus) that have a direct and positive effect on plant growth and SOM, a determinant for soil quality. SOM, the precursor to soil sustainability, is an important part of the labile (reactive) pool which also determines the ability of soil to retain water and nutrients. Studies have shown that soil organic C and N, and microbial biomass C and N increased or at least remain stable under grazed pastures compared to soils in croplands (Grace et al., 1998). Grazed pastures provide a quick way to build carbon by growing perennial plants continuously throughout the year and minimize disturbances to soil compared to cropping (Kirkegaard et al., 2007). Land management practices such as conservation agriculture (minimum or no tillage, crop rotation, cover crops, mulching, integration of livestock with cropping, and the introduction of legumes) that influence SOM are important for enhancing soil physical and chemical characteristics and minimizing livestock impacts on land degradation.

4.4 Grazing and Soil Quality

Soil physical, chemical, and biological properties collectively determine the quality of the soil and are affected by grazing. In New Zealand, after an extensive soil quality measurement program, the total C, total N, mineralizable N, pH, Olsen P, bulk density, and macroporosity were considered for regional soil quality assessment (Sparling et al., 2004). The assessment showed that while pasture soils are less vulnerable to adverse impacts on soil quality than crop lands, they are not completely resilient to withstand the negative impacts of the feed and grazing pressure (Cuttle, 2009; Sparling et al., 2004). Poor and uncontrolled grazing increases the loss of vegetative cover due to trampling and grazing plants too close to the soil (Nguyen et al., 1998). This weakens the root systems and compacts the soil leading to reduced soil quality. The degradations to soil quality can increase the soil erosion and nutrient losses from pastures and can, in turn, pollute surface waters. For example, in Africa overgrazing of marginal lands resulted in soil erosion and degradation (Lal, 1990). Overgrazing in livestock management is the main cause of soil degradation in Africa (50%), in the South Pacific, and in Australia (80%) (http://www.goodplanet.info/eng/Pollution/Soils/Soil-degradation/(theme)/1662). In temperate regions, intensively managed grazing systems can result in decreased pasture yield, biodiversity losses, reduced soil weight-bearing capacity, soil quality, soil erosion, and overland flow (CAST, 2002). Improving grazing management (reducing stock numbers or changing grazing period from long to short duration) retains complete vegetative cover, increased organic matter of the soil leading to improved soil structure that will allow greater water infiltration. This will allow more water used for plant growth rather than running off the land (Bilotta et al., 2007; Kemp and Michalk, 2005).

4.5 Carbon and Nutrient Cycling

Large areas occupied by grazing lands, their climate, and soil diversity, and the potential to improve their use and productivity make grazing lands important for sequestering C, mitigating emissions, and other aspects of global climate change (Lal and Follett, 2009). However, soil C stocks have shown both increases and decreases in response to different levels of grazing and are complex and depend upon various factors including pasture growth, pasture utilization by grazing animals, environmental factors, and the inherent soil properties (Schuman et al., 2002; Zhou et al., 2010). Grazing can increase the net primary productivity and dung and urine deposition on pasture lands. As a small proportion (15–25%) of the ingested nutrients are essentially retained in animal products and the remaining excreted to soil as dung and urine, animal wastes are an important soil fertility pathway in grazed pastures (West and Mallarino, 1996). Grazing enhances nutrient cycling; stimulates root respiration, root exudates, and carbon allocation below ground (Wardle and Bardgett, 2004); and increased soil C stock (Schuman et al., 2002). Productive and sustainable grazing lands with high-quality vegetation and soils provide high rates of C sequestration and low levels of carbon dioxide (CO_2) emissions. The total soil C sequestration potential for US grazing lands is approximately 70 million tons of C/year. This figure represents a C store of about 1.6 times the size of the CO_2–C emission from all other US agriculture and about 4.4 times the CO_2–C emission for all grazing lands (Lal and Follett, 2009). Similarly a review of 115 studies in 17 countries mainly with nontropical grasslands showed that a general improvement in soil carbon content (more than 70% of studies) and the improvement are influenced by management practices that include fertilizer application, grazing management, irrigation, and other conservation practices (Conant et al., 2001).

Grazing animal behavior directly influences the distribution of nutrients to various landscape positions. Animals may graze in one area but move to another area to rest or to drink water. Animal excreta (dung and urine) may thus be plentiful in the resting area and around a watering place than in the grazing area, resulting in a net transfer of nutrients and thus improving the fertility of soil in the resting area (Saggar et al., 1990). Grazing promotes nutrient cycling through rapid breakdown of organic matter into smaller particles in the system, so organic matter is available more readily for bacteria and fungi. Microorganisms use the organic matter as an energy source and can release nutrients back into the soil for plant uptake (Bloem et al., 2006). Thus nutrient management under grazing is highly complex, as a result of the uneven distributions of nutrients in pastures (Nguyen et al., 1989), the difference in nutrient element requirements for pasture growth, and the difference in mobility between nutrients and the tendency of nutrient (especially N) to escape from the system (Fig. 2).

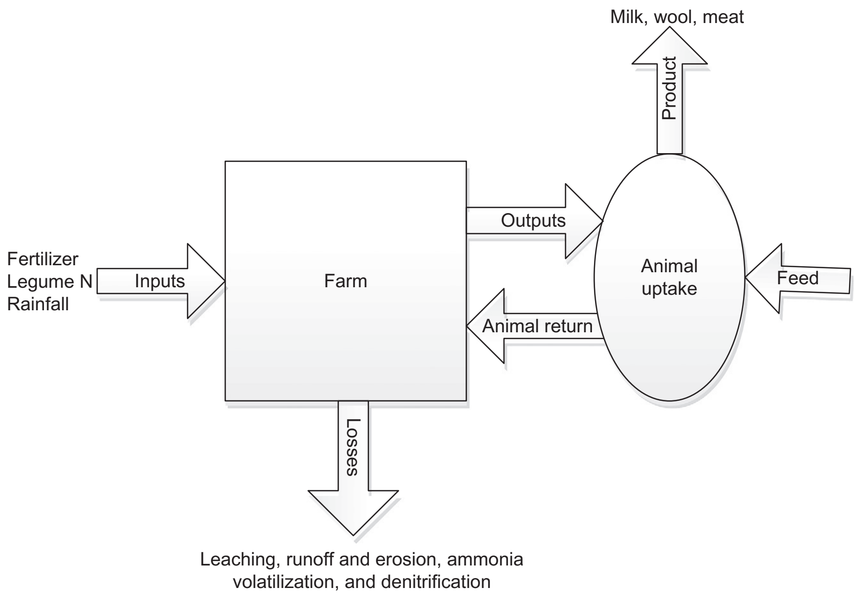

Fig. 2 Nutrient cycling in a grazed pasture system.

In the absence of fertilizer or outside manure inputs, continuous grazing will cause a gradual decrease of plant nutrients in the soil (Lory and Roberts, 2000). In concentrated dairy and other livestock operating systems, nutrient return from animals fed with high concentration of grain and protein supplements can be substantial. Winter feeds also form a substantial input into the pasture nutrient budget when animals are fed with hay while being kept on pasture. A dairy cow, for example, can return approximately 67 kg N, 14 kg P, and 10 kg K/year through dung and urine (Lory and Roberts, 2000). A uniform distribution of nutrients throughout a paddock is required for the productive use of nutrients for plant and animal growth. However, studies have shown that urine spots occupied 16.7% of the pasture, while manure spots occupied 18.8%, following 1108 grazing days per hectare (Dalrymple, 1994). Such an uneven distribution of dung and urine creates a large spatial variation in soil nutrients leading to large losses of nutrient through leaching, surface runoff, and in the case of N with volatilization and denitrification. Short period rotation of animals may help to reduce the spatial variability in the distribution of urine and dung and may reduce losses of nutrients from the production system. Grazing management practices which ensure uniform distribution of excreta can help improve both soil quality and fertility and reduce losses from the system.

5. NITROGEN AND PHOSPHORUS LOSSES UNDER LIVESTOCK PRODUCTION

Livestock production systems are one of the major causes of human-induced global N and P cycles (Bouwman et al., 2013). At the beginning of 1900, N and P input and output in livestock production were generally balanced and ever since the inputs were increasing compared to outputs (Bouwman et al., 2009).

5.1 Nitrogen

In livestock production system, about 55–95% of N intake is normally returned to the land as dung and urine. Generally at a global scale about 40–50% of the amount of N voided through dung and urine is collected from production areas such as paddocks, barns, and stables, and only 50% of this amount is recycled to crop land (Oenema and Tamminga, 2005). This translates to between 80 and 130 million tons of N annually which is either as large as or larger than the amount of fertilizer N used. Under intensive grazing management systems in which animals at a high stocking density are rotated through several paddocks at short time intervals (12–24 h), about 2% of the fecal N and 25% urinary N leached beneath the root zone (Stout et al., 2000a,b). The high rates of N fertilizer application and/or biological N fixation in these grazing systems and the uneven recycling of N through urine and dung in pastures could increase N leaching and contributing to increased groundwater nitrate levels. High nitrate leaching was observed when a severe drought followed good growing conditions, causing legume nodules to die and release nitrogen into the soil (Stout et al., 2000a).

Nitrogen losses from sheep-grazed pastures in New Zealand showed that overland flow and interflow losses ranged from <1 to 19 kg N/ha/year (Melland, 2003; Parfitt et al., 2007; Ridley et al., 2001, 2003) and losses through deep drainage ranged from <1 to 50 kg N/ha/year, with concentrations generally ranged between 2 and 25 mg N/L (Cuttle et al., 1992; Parfitt et al., 2007; Ridley et al., 2001, 2003; Ruz-Jerez et al., 1995). Data on N flows in grazed dairy pastures in the last four decades clearly showed that N excreted by animals, particularly urine, is the most important determinant of both deep drainage and runoff (Monaghan, 2009). N losses through subsurface drainage ranged from 19 to 150 kg N/ha/year as nitrate in New Zealand (Ledgard et al., 1999; Monaghan and Smith, 2004;

Monaghan et al., 2005), 59 to 194 kg N/ha/year in United Kingdom (Haygarth et al., 1998), and 18 to 65 kg N/ha/year in Ireland (Watson et al., 2000). The above results from both sheep and dairy farms show that N losses from sheep-grazed pasture are less than that from the cattle-grazed pastures. This reflects the nature of sheep-grazed pastures (lower stocking rates, soil fertility, and the return of N in dung and urine) and the difference in the amount and size of urine and fecal patches (Haygarth et al., 1998; Haynes and Williams, 1993).

5.2 Phosphorus

Phosphorus management under grazing system represents a major challenge to agronomist, natural resource managers, and environmentalist (McDowell et al., 2004; Nguyen and Goh, 1992). While P is one of the major essential nutrients for a profitable livestock operation, the transport of P from grazing lands to surface water bodies and the high cost associated with commercial P fertilizers emphasized the reevaluation of phosphorus management strategies that optimize feed and livestock production with little or no negative impact on receiving waters (Sharpley et al., 1994). In most grazing systems (particularly those that receive animal manure), more P is applied to soil relative to N as the N:P ratio of dung and urine (4:1 to 5:1) does not match the crop/feed requirement (more P relative to N). Manure application rates are determined on the basis of N requirements of feed crops leading to higher P application. Soil P accumulation rates due to manure application in the United States and several European countries range from 8 to 40 kg P/ha/year with an average rate of 22 kg P/ha/year (Carpenter et al., 1998). Dissolved P from manure and P fertilizers lie on the soil surface. When the amount of P in soil exceeds the ability of soil particles to bind onto it and crop P requirements, the excess P can readily be dissolved and transported by runoff water during rainfall (Heathwaite et al., 2000).

Phosphorus losses from grazed pastures are closely related to overland flow and tied to both surface runoff and sediment movement (Leinweber et al., 2002). Therefore, P losses from pasture with exposed surface soils are greater than properly managed pastures with a limited amount of bare soil. In addition, P application method, rate of application, timing, and form of P added as fertilizer and manure, and erosion and runoff potential influence P losses from grazed pasture (Kleinman et al., 2011; Sharpley et al., 2001). Estimations from grazing system in New Zealand have shown that fertilizer, dung, pasture plant, and soil contributed to approximately 10%, 30%, 20%, and 40% of phosphorus losses (McDowell et al., 2006). In Australia, P losses from grazed paddocks are three times that of ungrazed pasture (McDowell et al., 2006). In the United States, seasonal P losses through surface runoff ranged from <0.1 to 1.28 kg P/ha with P concentration ranged from 0.45 to 2.51 mg/L in runoff water (Owens and Shipitalo, 2006). Compared to surface runoff losses, P losses through subsurface flow are much lower and ranged from 0.024 to 0.082 kg P/ha (Owens and Shipitalo, 2006).

Phosphorus losses from livestock production are generally low unless management practices (e.g., overgrazing) or soil and topographic conditions (e.g., steep slopes) are conducive for accelerated erosion (Schlesinger et al., 1996). While environmentally significant, the P losses from livestock production systems via surface and subsurface flows are rarely significant in terms of productivity. The lower P losses from sheep-grazed than cattle-grazed pastures reflect the extensive nature of sheep-grazed pastures (lower stocking rates, soil fertility, inputs of fertilizer, and supplementary feed) compared to that in cattle farming. Additionally, less urine and dung deposition, from sheep compared with cattle, may also decrease P losses. Therefore, opportunities exist for reducing P losses from livestock production systems through monitoring soil P reserves, the type of stock, and stocking rates. Understanding local capabilities and farming technologies are fundamental for developing strategies (e.g., identification of critical sources areas for different grazing and P fertilizer management, set-aside riparian zones, and fencing grazing livestocks from streams) to effectively use and manage P in livestock production. The focus will be on to determine how these strategies can affect soil fertility, P use efficiency, and losses to surface waters (Lucci et al., 2010; McDowell et al., 2011).

6. LIVESTOCK PRODUCTION AND GHG EMISSION

Currently, there are a large and growing number of published reviews, and results of field research are available on GHG emissions from livestock production (Basset-Mens and van der Werf, 2005; Casey and Holden, 2006; FAO, 2006a; Garnett, 2009; Lovett et al., 2006). The existing information clearly showed that most of the GHG emission from livestock occurs during the production with little GHG emission during postproduction. There are a number of publications worldwide for estimating GHG emission from livestock production. For example, all developed nations have implemented their greenhouse accounting as part of United Nations Framework Convention on Climate Change (UNFCCC) and livestock is part of this accounting. Agriculture accounts for 84% and 52% of global nitrous oxide (N_2O) and methane (CH_4) emissions, respectively, and makes up about 14.5% of anthropogenic GHG emission (Denman et al., 2007; US-EPA, 2006). Within agriculture, the livestock sector accounts for an estimated 9% of global CO_2 emission, 65% N_2O emission, and 35–40% CH_4 emission. N_2O emissions from soils and CH_4 from enteric fermentation constitute the largest sources, being 38% and 32% of total non-CO_2 emissions from agriculture, respectively (US-EPA, 2006). Such emissions represent a significant proportion for some developed countries such as New Zealand, Ireland, and the United Kingdom (Moran and Wall, 2011). In Australia, direct livestock emission of GHG accounts for 10% of total GHG emission (http://www.agriculture.gov.au/climatechange/australias-farming-future/livestock-emissions). Nitrous oxide emissions from agricultural soils are projected to increase 37%, and enteric livestock CH_4 emissions are projected to increase by 23% and manure CH_4 and N_2O to increase by 15% and 19% by 2030 compared with 2005 levels (US-EPA, 2012). Most of this increase in CH_4 and N_2O emission occurs in Asia, Latin America, and Africa, because of increased demand for livestock products due to changing diet preferences (FAO, 2006a,b). Baseline estimates of GHG emission for 2000, 2010, and 2020 under livestock management (mainly through enteric fermentation and manure management) for some countries and regions are provided in Table 4. Of the total agricultural emission, grazed pasture systems contribute 20% of CH_4 and between 16% and 33% of N_2O emissions globally (Clark et al., 2005).

6.1 Nitrous Oxide

Atmospheric concentrations of N_2O have increased significantly since 1860 with livestock production being the major contributors (Wolf et al., 2010). Globally, livestock-related N_2O emission was estimated at 1–2 million tons N_2O–N per annum. Relatively a small amount of N_2O is produced directly by livestock. A majority of N_2O emission is related to production of livestock and postproduction management. The storage and treatment of livestock manure produce N_2O through a combination of nitrification and denitrification (Fig. 3). As aerobic reactions and anaerobic processes are important for N_2O production, systems that involve aerobic management are conducive for N_2O production.

Table 4 Baseline GHG Emission From Livestock Management (Million Tons CO_2 Equivalent)

	Year		
Country/Region	**2000**	**2010**	**2020**
Africa	271	332	395
Australia/New Zealand	91	93	94
Brazil	222	263	297
China	313	392	470
European Union	48	54	58
India	224	260	286
United States	171	173	171
World	2220	2548	2867

US-EPA, 2006. Global anthropogenic non-CO_2 greenhouse gas emissions: 1990–2020. EPA 430-R-06-003. United States Environmental Protection Agency; June 2006. Washington, DC. http://www.epa.gov/nonco2/econ-inv/downloads/GlobalAnthroEmissionsReport.pdf (accessed 26 March 2007).

Stepwise denitrification

$$NO_3(\text{solution}) \rightarrow NO_2(\text{solution}) \rightarrow NO(\text{gas}) \rightarrow N_2O(\text{gas}) \rightarrow N_2(\text{gas})$$

The reduction of NO to N_2O is mediated by nitric oxide reductase

$$2NO(\text{gas}) + 2H^+ + 2e^- \rightarrow N_2O(\text{gas}) + H_2O$$

The reduction of N_2O to N_2 is mediated by N_2O reductase

$$N_2O(\text{gas}) + 2H^+ + 2e^- \rightarrow N_2(\text{gas}) + H_2O$$

Stepwise nitrification

$$NH_3(\text{solution}) \rightarrow NH_2OH(\text{solution}) \rightarrow N_2O(\text{gas})$$

Stepwise nitrifier denitrification

$$NH_3(\text{solution}) \rightarrow NH_2OH(\text{solution}) \rightarrow NO_2^-(\text{solution}) \rightarrow NO(\text{gas}) \rightarrow N_2O(\text{gas}) \rightarrow N_2(\text{gas})$$

Fig. 3 Nitrification and denitrification processes in agriculture soils producing nitrous oxide. The process of nitrification and nitrifier denitrification overlaps. *Adapted from Pilegaard, K., 2013. Processes regulating nitric oxide emissions from soils. Philos. Trans. R. Soc. B 368, 20130126; http://dx.doi.org/10.1098/rstb.2013.0126 and Wrage, N., Velthof, G., van Beusichem, M., Oenema, O., 2001. Role of nitrifier denitrification in the production of nitrous oxide. Soil Biol. Biochem. 33, 1723–1732. http://dx.doi.org/10.1016/S0038-0717(01)00096-7.*

Nitrous oxide emissions from grazed pastures are associated with C and N from the deposition of dung and urine to the soil by grazing animals and are highly variable with time and space (Abdalla et al., 2009; Bolan et al., 2004; Saggar et al., 2007). The uneven distribution of dung and urine, soil water status, fertilizer application, soil N mineralization, soil–plant interaction, and climatic conditions are important factors contributing to such variability (Bolan et al., 2004; Liebig et al., 2005; van der Weerden et al., 2010; Wolf et al., 2010). N_2O emissions from grazed pastures ranged from zero in arid and infertile regions to 15 kg N/ha in soils with well-fertilized, wet, and anaerobic conditions (Bolan et al., 2004; de Klein et al., 2009).

Between 1990 and 2008, N_2O emission from grasslands decreased by 3.7%, while it is increased by 18.75% in livestock manure management in the United States (US-EPA, 2010). In Australia, agricultural activities in general accounted for 76% and soil management accounts for 17% of the total N_2O emission, respectively. Livestock production accounts for >43% of anthropogenic N_2O emission which is more than the global average of 28% (DCC, 2010). In the European Union, emission from agriculture is the largest source of anthropogenic N_2O emission accounting 78.5% and livestock production (manure and soil management including fertilizer application) accounts for 51% agricultural N_2O emission (EEA, 2015). Nitrous oxide emissions from livestock production decreased by approximately 23% during the period 1990–2013. This is associated with the adoption of a number of climate-specific environmental policies, reduced use of N fertilizer, and soil management practices in the European Union (EEA, 2015; Smith et al., 2007a). In China, the mean annual N_2O emission from grasslands account for approximately 9.56 million tons/year. However, in recent years, these grasslands have been intensively developed, so future estimates should include land use change, spatial and temporal distribution of grazing, and fertilizer management (Zhang et al., 2010).

Major processes that control N_2O emission from grazed pastures include (1) soil nitrification and denitrification rates; (2) the ratio of the end products of denitrification, i.e., N_2O vs N_2; and (3) the diffusion of N_2O in the soil. A number of factors affect these processes in the soil and control denitrification. These include (1) fertilizer application, (2) return of dung and urine from grazing animals, (3) frequent wetting and drying of soils, (4) animal treading, and (5) soil temperature. However, the interaction between these factors is complex, and as a result, it is impossible to distinguish the effect of various factors on N_2O emission.

Fertilizer application to pastures increased denitrification rates compared to unfertilized pastures and denitrification increase with increasing fertilizer application rate. In managed pastures, fertilizer application contributes up to 25% N losses through denitrification (Bolan et al., 2004; Ledgard et al., 1996; Prasetstak et al., 2001). Annual denitrification losses were estimated between 6 and 15 kg N/ha/year for south-east Australian dairy pastures receiving fertilizer (Eckard et al., 2003). In grazed pasture systems in New Zealand and Australia that are characterized by year-round grazing and limited use of N fertilizer (on average of 100–150 kg N/ha/year), N_2O emissions from dung and urine patches generally contribute about 50–60% of the total N_2O emissions and direct emission from fertilizer accounts between 10% and 15% (de Klein et al., 2001). In high N fertilizer inputs (over 200 kg N/ha/year) dairy systems in Europe that are characterized by animal housing for 5–6 months per year, direct deposition of excreta contributed to less than 50% of the total N_2O emissions. Fertilizer inputs and N applied as manure or slurry contribute up to 35% and 15% of the emissions, respectively (Schils et al., 2005). The consumption of oxygen by soil microorganisms in urine and dung affected area and an increase in the availability of carbon can contribute to increased N_2O emissions from dung and urine patches (Oenema et al., 1997). In addition to direct emission, indirect emission of N_2O from fertilizer, dung, and urine represents about 20–30% of the N_2O emissions (Schils et al., 2005).

Soil wetting and drying influence denitrification in pasture soils and unraveling the relationship is very complex. When the soil is fully saturated, soil aeration is restricted, the redox potential in the soil decreased, and the rate of denitrification increased. However, as the nitrate concentration in saturated soils is low due to slow nitrification, the overall losses of N through denitrification may be lower under such conditions (Akatsuka and Mitamura, 2011). Above the soil water level of field capacity, denitrification increased with increasing water content (de Klein and van Logtestijn, 1996). Studies conducted in clay loam pasture soils in United Kingdom have shown that denitrification increased with increasing water content (0.46, 0.92, and 3.38 kg N_2O–N/ha/day at 63%, 71%, and 84% soil water, respectively) (Abbasi and Adams, 2000).

Animal treading stimulates short-term denitrification in pasture soils as it causes a temporary reduction in soil aeration through soil physical damage and reduced soil N utilization due to reduced plant growth and defoliation (van Groenigen et al., 2005). In pastures with high stocking rates (4.5 cows per 100 m^2) denitrification reached 52 g N/ha/day, 8 days after severe treading compared to 2.3 g N/ha/day compared to no treading (Menneer et al., 2005), suggesting the influence of grazing management on denitrification. Integration of cropping with livestock system may provide some opportunities to reduce N_2O emissions as N inputs are controlled by efficient use of crop residues. As considerable body of information is available on N_2O emission from livestock is available, it is important to look at the mechanisms of N_2O emission as it provides information on pathways of denitrification and develop management practices to reduce the emission in livestock production.

6.2 Methane Emission

As discussed earlier, intensive production system to meet the increasing demand for livestock products led to a high-density livestock systems leading to strong localized methane (CH_4) emissions. CH_4 emission from livestock production can derive from (1) enteric fermentation, (2) anaerobic decomposition of organic materials in livestock manure, (3) manure deposited on fields and pastures, or handled in a dry form, and (4) liquid manure management systems, such as lagoons and holding tanks (US-EPA, 2010). Currently the livestock sector contributes to 35% of global anthropogenic CH_4 emission. Based upon the growth in livestock numbers, global CH_4 emission could increase by up to 60% by 2030 (FAO, 2003). However, improved livestock management practices (feeding practices and manure management) could ameliorate such increase and recent forecast suggested an increase of around 21% CH_4 emission from enteric fermentation and manure management (US-EPA, 2010).

In the United States, CH_4 emission from livestock production accounted for approximately 33% of total CH_4 emission in 2008 (US-EPA, 2010). Between 1990 and 2008 livestock CH_4 emission increased by 14%, while the total methane emission decreased by 7.5%. Methane emission increased by 6.3% by enteric fermentation and close to 50% by manure management during the period between 1990 and 2010 (US-EPA, 2010). In the European Union, livestock production contributes to approximately 4.2% of the total CH_4 emission (source: *EEA Technical report 4/2009*). Around 7.8 million tons of methane per year was produced in Africa in 2000 and this figure could likely to increase to 11.1 million tons per year by 2030. Methane emissions per tropical livestock unit (TLU, 250 kg bodyweight) can vary from 21 to 40 kg/TLU/year, depending on the production system and the region (Herrero et al., 2008). Methane emissions from livestock in South, Southeast, and East Asia were estimated to be about 29.9 million tons per year CH_4 in 2000. These emissions consisted of 25.9 million tons CH_4

from enteric fermentation and 4.0 million tons CH_4 from livestock manure management systems. India had the greatest production of CH_4, with 11.8 million tons CH_4 from livestock, primarily from cattle and buffaloes. China was also a high-emission country, producing about 10.4 million tons CH_4. The total methane emissions from livestock increased by an average of 2% per annum from 1965 to 2000 in CH_4, indicating the significance of livestock production on climate change (Yamaji et al., 2003).

Studies have shown that more than 95% of global CH_4 emission originates from enteric fermentation and manure management (Smith et al., 2007a), and CH_4 emission from soil and plants in grazed pastures is minor. Although it is possible that CH_4 emission may occur in micro-anaerobic soil conditions under grazing, well-drained pasture soils are generally considered as sinks for CH_4 (Nicol et al., 2003). A number of factors, including soil water content, oxygen availability, organic matter content, temperature, fertilizer, and microbial inhibitors, affect CH_4 emission from grazed pastures (Le Mer and Roger, 2001). It has been claimed that pasture plants release CH_4 (Keppler et al., 2006); however, such claims have been challenged based on results which showed that CH_4 release from pasture plants could be 3 kg/ha/year and is less than 3% of CH_4 released from enteric fermentation (Parsons et al., 2006).

7. ISOTOPIC AND NUCLEAR TECHNIQUES FOR ASSESSING SOIL–CROP–LIVESTOCK INTERACTION

Assessing fluxes of water, nutrient, GHGs, and their sources under livestock production is difficult to establish by conventional techniques and require assumptions. A number of isotopic techniques that include the use of stable isotopic ratios of oxygen ($^{18}O/^{16}O$) and hydrogen ($^{2}H/^{1}H$) in water and water vapor can be useful to separate evaporation from transpiration because both processes have different effects on these isotopic ratios in water (Jasechko et al., 2013). Recent developments in isotopic methods, such as $\delta^{15}N$–$\delta^{18}O$ nitrate-specific isotopic techniques (McIlvin and Casciotti, 2011; Verburg and Kendall, 2013), the $\delta^{18}O$ of phosphate (Paytan and McLaughlin, 2011; Tamburini et al., 2010), NO_3^- source apportionment using $d^{15}N-NO_3^-$ and $d^{18}O-NO_3^-$ (Xue et al., 2013), compound-specific isotope methods (Gibbs, 2008), and the use of fallout radionuclides, have enhanced our ability to use isotopic tracing approaches to identify sources, pathways, and transformations of nutrients and sediments in agriculture including livestock production systems.

In recent years, measurement of surface soil moisture by the cosmic-ray neutron probe has gradually attracted more attention to soil and water scientists (Desilets et al., 2010) including grazing systems. The intensity of the fast neutrons above the ground is sensitive to changes in water content, largely insensitive to soil chemistry and inversely correlated with hydrogen content of the soil. By this passive, noninvasive, and intermediate scale measurement, soil moisture at a horizontal scale of around 40 ha and depths of 12–76 cm can be inferred. The large footprint of this tool makes this method suitable for weather and short-term climate forecast initialization and for validation of soil moisture inversed from satellite sensors (Zreda et al., 2012). However, there are also many problems to be solved in cosmic-ray neutron method, such as surface water effect during irrigation when cosmic-ray soil moisture was much higher and not consistent with the real situation. Also how to deal with the impact of other external factors (such as the air humidity, magnetic field changes, other water sources, and the change in water content in the growing plant) impacts on the neutrons (Jiao et al., 2014). Once the technique is calibrated to address these external factors affecting the measurement, this technique plays an important role for area-wide soil water assessment. Compound-specific stable-isotope analysis (CSIA) has facilitated the assessment of sources and transformation processes of pollutants in agricultural landscapes (Gibbs, 2008). CSIA is in transition from a research tool to an applied method that is well integrated into comprehensive plans for management of agricultural landscapes. The technique is particularly useful for sediment tracing and relies on the ability to characterize and discriminate material from different sediment source areas using a suite of natural tracer properties (Gibbs, 2008). The most important assumption made with CSIA is that sediment fingerprint properties are not transformed during transportation to downstream or storage in the river or stream system. Once calibrated, CSIA may be useful in identifying sources of N from grazing and nongrazing systems. Measurement of changes in the ^{15}N isotopic signatures of various components (nitrate, nitrite, N_2O, and N_2) during nitrification and denitrification processes could also help identify the main process and develop management strategies for reducing N_2O emissions (Mander et al., 2003).

8. CONCLUSION

While the livestock sector is significantly important for global food security and economic activity, the interaction between livestock production and the environment is both a political and a biophysical threat to the sustainability of the industry. As the intensity of livestock production increases to feed the increased global population, appropriation of more resources to support the expansion is required and this will put enormous pressure on ecosystem beyond their capacity leading to the collapse of the ecosystem and seriously affect services provided by the ecosystem. Additionally, in grasslands and permanent pastures, changes in livestock management by enabling maximum ground cover, minimal soil compaction, and efficient nutrient utilization are difficult to achieve as the intensity of livestock production increases leading to more land and water quality degradation. Further, the increased livestock intensification and demand for limited water resources will force stakeholders to think carefully about the impacts of livestock production on water resources.

Without any corrective measures livestock intensification leads to (1) large areas of grazing land with high concentrations of N and P polluting surface and ground waters, (2) more native forests will be cleared for feed crop production leading to the loss of natural habitats and biodiversity; as more forests are cleared the natural hydrology of the landscape changes leading to unpredictable discharge and flooding, (3) anthropogenic GHG emission will increase leading to climate change and extreme weather events, and (4) degradation of arid and semiarid land will continue with marginalization of rural poor. However, there is optimism toward a sustainable livestock industry by addressing the conflicting demands for livestock production and environment. This sustainability can be achieved through innovative new technologies that will reduce the environmental impacts of livestock production and improve efficiency in the sector. Encourage efficiency through adequate policies, institutional support, and market prices for the resources being used in livestock production is important for the sector to expand and play a major role in global hunger, nutrition, and poverty reduction.

Proper incorporation of livestock manure into forage production systems would be an effective way of recycling manure to improve soil fertility and quality and reduce impacts of livestock management on water quality degradation. Forage production removes and recycles more nutrients from the soil than other crops especially when plants of high biomass yield with relatively high N uptake capacity, tolerance to wet soil conditions, prolonged vegetative growth, and tolerance to frequent harvest are used. This will maximize manure nutrient utilization and reduce N and P movement to surface and ground water. When using livestock manure for forage production, careful consideration will be given to the N and P concentrations of manure. Integrating cropping with livestock production system reduces overall fertilizer consumption leading to reduced losses of N and P from the system compare to stand alone cropping or livestock production systems.

Stocking density in intensively managed pastures is an important factor controlling the extent and magnitude of the environmental impacts of livestock production. At low stocking density the number and frequency of hooves decreased leading to enhanced nutrient cycling and biodiversity, and at high stocking density, damage to pasture may occur leading to soil erosion, runoff, and water quality degradation. Reducing stocking density may be one option to reduce the environmental impacts. However, at low stocking density the economic return may be affected. The challenge is to optimize the stocking density taking into consideration factors such as soil texture, topography, and the drainage system that reduces the environmental impacts of grazing without compromising the economic return of small farm holders. The development of GHG mitigation strategies and the ability to understand and adapt to changing environment are critical for sustaining livestock production system and reducing its GHG emissions. Continuous monitoring of the extent of additional livestock development and their impact on fertilizer and water use, water quality, and GHG emission are important to sustain the industry.

The livestock industry, though of modest economic importance, has an overwhelming social importance in almost all developing countries as it is important for supporting rural population through year-round income and human nutrition. Over the years the economic importance of livestock production has been given widespread attention. Even though it is possible to reduce the environmental impacts of livestock production at a reasonable cost, it has not been given priority in the past. Adequate institutional support and political willingness are important to reduce environmental impacts including GHG emissions and land and water quality degradation. A sustainable and profitable livestock system is possible through improved technologies, practices, and policies that promote incentives to stakeholders and balance the trade-off between sustainability and profitability of the system. Further research focus can be directed toward integrating crops with livestock and incorporating silvipastoral systems in livestock production that can improve the system's environmental and climate footprints. Identifying sources of N_2O and CH_4 emissions from soils could help develop management plans that control source availability and overall GHG emissions reduction under livestock production.

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