Chapter 10
Intensification in Pastoral Farming: Impacts on Soil Attributes and Gaseous Emissions

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10.1 Introduction

Grasslands worldwide cover about 25% of the earth's surface, occupy 117 million km^2 of vegetated land and provide forage for over 1,800 million livestock units (Saggar et al. 2009a). They are also one of the key contributors to potent non-carbon dioxide (CO_2) greenhouse gases (GHGs) (Clark et al. 2005). Methane (CH_4) produced by the fermentation of organic matter in an anaerobic environment has a global warming potential (GWP) of at least ~25 times greater than CO_2 (Shindell et al. 2009). The GWP of nitrous oxide (N_2O), which is produced in pastoral soils from mineral nitrogen (N) originating from dung, urine, biologically fixed dinitrogen (N_2) (BFN), applied fertiliser and mineralisation of soil organic matter, is even greater (310). Livestock production is responsible for 18% of global GHG emissions from all human activities measured on a CO_2-equivalent basis (Steinfeld et al. 2006). Intensification of managed pastoral soils affects GHGs emissions and modifies soil properties that have wider environmental impacts on water and air quality. The emissions per unit of milk production are highest in developing regions and least in North America and Europe, and are higher in grazing systems than mixed systems (FAO 2008). Although intensification can produce less GHGs per

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unit of output, intensification can impact negatively on soil quality, biodiversity and eutrophication of water. Moreover, "Compassion in World Farming" – the farm animal welfare charity, considers that intensification is a deeply flawed strategy from the point of view of halting climate change and from environmental and animal welfare considerations (FAO 2008).

Pastoral landscapes are generally managed at the individual farm level, but the impact occurs at catchment level. Farmers have several goals – economic, environmental and lifestyle – that will affect how they manage their land and resources. In countries such as New Zealand, where the economy relies heavily on export income from pastoral agriculture, farmers face pressure to increase efficiency of production to maintain their financial and economic viability, as well as their position as low-cost producers in an internationally competitive market. One way of achieving this is to increase an efficient use of inputs to stimulate a larger increase in returns, and hence an increased output–input ratio. Present pastoral farming trends in New Zealand show that the sector is growing and using higher inputs, including fertiliser, energy, water for irrigation and capital to produce more output from the same area of land (PCE 2004). Reviews by Saggar et al. (2004b, 2009a) and Bolan et al. (2009) provide some data on major changes in New Zealand dairy industry. We can also assume that intensification will continue to occur to meet ever-increasing demands for food, even within landscapes already farmed relatively intensively.

Pastoral agriculture has traditionally focussed on outputs of products, i.e. meat, fibre and milk. However, farming systems have other outputs/effects, e.g. loss of nutrients to water and other impacts on soil properties, and GHG emissions. Pastoral farms can be regarded as forage supply platforms, so more forage produced per unit area is a primary aim. The environmental costs associated with intensive livestock farming (including confined livestock operations) are the disposal of waste products that may cause soil, air and water pollution, increased disease risks to animals and humans, the reduction in biodiversity and increased GHG emissions.

In this chapter, we describe intensification of pastoral agriculture and address its impacts on physical and chemical soil attributes of intensively managed pasture land, with particular emphasis on temperate-grazed pasture systems. We investigate how increased chemical inputs can affect the productivity and environmental quality of these pastoral soils. We describe how intensive management of the pastoral system influences emissions of GHGs. We also explore options for reducing the negative impacts of intensification, and identify current gaps and limitations for developing future sustainable pastoral systems management strategies while maintaining productivity, profitability and the environment. As only a brief description of intensification impacts on pastoral farming is given in this chapter, the reader is referred to appropriate reviews that provide in-depth coverage of relevant topics (Bilotta et al. 2007; Bolan et al. 2004; Carroll et al. 2004; Cuttle 2008; DeKlein and Eckard 2008; Drewry 2006; Kemp and Michalk 2005; Kurtz et al. 2006; Ledgard and Luo 2008; Luo et al. 2010; Oliver et al. 2005; Saggar et al. 2004b, 2009a).

10.2 Intensification of the Pastoral Land Use

Pastoral intensification is a broad term and includes increases in the level of inputs such as fertiliser, stocking rate, irrigation, chemicals, plant and animal germplasm, machinery, labour, and biotechnologies. Intensification here refers to any practice that increases productivity per unit land area at some cost in labour or capital inputs. Intensification of pastoral land use may be considered as a system to feed the world while avoiding the Malthusian outcome. But this unprecedented success has come at a large cost both to the environment and to human health. Responses to different inputs of intensification can have different consequences, e.g. fertiliser application may increase, decrease or have no effect on soil carbon (C) content, while increases in stocking rate may decrease soil C or may have some positive effect on soil C.

In more humid regions including Australia, New Zealand, Europe and parts of North and South America, most pastoral land is intensively managed with substantial inputs of N fertiliser. While pasture production commonly increases with increasing rate of N application, N use efficiency decreases (McKenzie et al. 2006). Pasture generally responds linearly to N application rates up to 200–400 kg N ha^{-1} $year^{-1}$ (Whitehead 1995; Sun et al. 2008). Higher inputs of N fertiliser can result in a large N surplus (i.e. N inputs–N outputs in products). For example, there have been N surplus of 150–250 kg N ha^{-1} $year^{-1}$ in highly productive dairy farm systems in the Netherlands and northern Germany (Rotz et al. 2005), mainly from excessive application of animal manure/excretal deposition. N surpluses in these intensively managed pasture systems are likely to exacerbate N losses to waterways and the atmosphere. The environmental effects of NO_3^- leached to groundwater and other waterways and the potential damage to soils are a major concern to the farming industry, the scientific community and the society. In New Zealand, the declining water quality of Lake Taupo (Vant and Huser 2000), the Rotorua Lakes and algal blooms occurring in Lake Rotoiti has been linked to the land use within the catchment. Environment Waikato data suggest the quality of about 10% of the groundwater in the livestock farming area of the region is below World Health Organisation drinking water standards (Annon 2005). Thus, excessive N additions can contaminate pastoral ecosystems and alter both their ecological functioning and the living communities they support. Another example here is the large dead zone in the Gulf of Mexico, which is a direct result of nutrients and agrochemical run-off from intensively managed agricultural land in the USA via the Mississippi river (McIsaac et al. 2001).

Recent trends in intensive pastoral land use in New Zealand include higher stocking rates and stocking densities, increased use of fertilisers and agricultural chemicals, and increases in irrigation use. In the past two decades, New Zealand pastoral farming has doubled milk production from dairying and, despite a one-third decline in ewes, lamb meat production has increased by 10% (Woodford 2006; Bolan et al. 2009). More intensive pastoral farming (increase in stocking rates or more livestock numbers per hectare) and more productivity per animal (such as increased milk production per cow, or higher lambing percentages and carcass weights) resulted in 38% increase in production between 1990 and 2003. Dairy cow numbers have almost doubled (from 2.92 million in 1981 to 5.22 million by 2006). Between 1990 and 2005, there has been a sixfold increase in N fertiliser use from 0.05 to 0.31 million tons N (MfE 2007a). It is estimated that New Zealand animals annually void almost five times more N (1.5 million tons) than the N fertiliser input (0.31 million tons N) (Saggar et al. 2005). Intensification of pastoral land use has also led to a noticeable increase in the use of irrigation in drier regions to achieve high-producing pastures (MfE 2007b). The combined increase in fertiliser use and irrigation has increased environmental pressures on waterways and groundwater. A shift to more intensive farming in some New Zealand regions has adversely affected soil health (Betteridge et al. 1999, 2002; MacKay 2009; Sparling and Schipper 2002), increased GHG emissions (MfE 2007b), decreased indigenous biodiversity (Leathwick et al. 2003) and reduced freshwater quality in lowland waters and waterways (Quinn et al. 1997). This has raised concerns about ecological sustainability of New Zealand pastoral farming and continuation of its intensification in the future (MacLeod and Moller 2006).

Phosphorus (P) is the major nutrient limiting the growth of clover-based pastures in New Zealand, and superphosphate has been the major P fertiliser in use (Saggar et al. 1993). Pasture improvement can influence P fluxes in waterways and streams by increasing P transfer from the soil, applied fertiliser and animal excretal deposition. In most intensive livestock production areas around the world P inputs are in excess of requirements (Sharpley et al. 1998; OECD 2008). As P requirements for intensively grazed pastures are relatively high and New Zealand soils are naturally P deficient (Caradus 1994; Sinclair et al. 1996), New Zealand dairy farms use large amounts of P fertilisers; this has increased the potential for P loss to waterways (Monaghan et al. 2007). In Northern England, Withers et al. (2007) showed a direct link between upland pasture improvements by liming and P fertilisation and soil P accumulation, which doubled the transfer of dissolved inorganic P and particulate P but not suspended sediment to the drainage stream. In New Zealand, sulphur (S) fertilisers are also applied on intensively grazed pastures in addition to that supplied through the commonly used single superphosphate fertiliser. Total application rate can range from 60 to 100 kg S ha^{-1} year^{-1} for New Zealand dairy farms, and leaching losses of 40–70 kg S ha^{-1} year^{-1} as sulphate-S have been reported (Rajendram et al. 1998).

10.3 Nutrient Inputs and Dynamics in Grazed Pastures

Historically, fertiliser applications have greatly increased pasture and animal production on many grassland soils that are inherently deficient in nutrients. The annual amount of fertiliser nutrients used in a farm system and those recycled through the uneven deposition of animal excreta are the two key factors determining the nutrient surplus, their spatial and temporal heterogeneity, their potential for loss and, therefore, the nutrient-use efficiency. Additionally, in legume-based pastures atmospheric N input through biological N_2 fixation can also contribute to significant N inputs. The amount of biological N_2 fixation depends on a number of factors, including legume species, climatic and soil conditions, nutrient supply and grazing management (Menneer et al. 2004). Estimates of N_2 fixed by legumes (mainly white and subterranean clovers) in temperate pastures throughout the world range from 10 to 270 kg N ha^{-1} $year^{-1}$ (Ledgard 2001). Biological N_2 fixation generally decreases in intensive pasture systems as inorganic N supply to the legumes increases (Saggar 2004; Saggar et al. 2009a; Fig. 10.1).

In grazed pastures, the conversion efficiency of consumed N, P and S into product is low, and a substantial amount of N, P and S (from 70 to 85%) is recycled through the direct deposition of animal excreta. The low utilisation of these pasture nutrients reflects a simple feature of the pasture–animal relationship: in most situations, pasture plants require significantly higher concentrations of N, P and S to grow at optimal rates than is needed by the grazing ruminant for amino acid and protein synthesis (Haynes and Williams 1993). The proportion of N in the urine increases with increasing N content of the diet. In most intensive high-producing pasture systems, where animal intake of N is high, more than half the N is excreted as urine (Oenema et al. 1997).

In intensive dairying systems where winters are cold (e.g. northern Europe), housing for varying periods throughout the year is common in grazing systems. This means collection and application of large quantities of manure become critical for nutrient-use efficiency, as there are many opportunities and places for N compounds to escape from animal manure management systems.

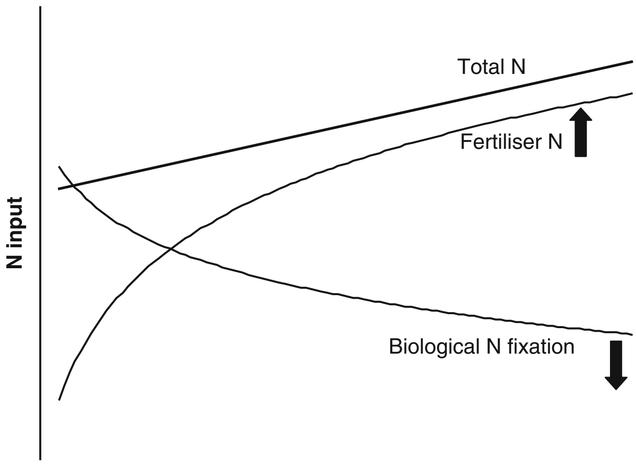

Fig. 10.1 Schematic representation of the influence of increased nitrogen (N) fertiliser application on biological nitrogen fixation (BNF) in legume-based pastures (adapted from Saggar 2004; Saggar et al. 2009a). The *x*-axis represents the changes in N contribution between BNF and fertiliser N

10.3.1 Nitrogen Transformation Processes in Grazed Pastures

The transformations and losses of N in managed grazed pastures have been reviewed (Bolan et al. 2004; Fig. 10.2). The N in excreta following deposition undergoes microbial mineralisation before it is released as the ammonium ion (NH_4^+) and NH_3 gas. N mineralisation is much faster from urine than from dung. N can be lost to the atmosphere by NH_3 volatilisation, or converted to nitrate (NO_3^-) through the nitrification process by nitrifying bacteria. Nitrate can then be leached and denitrified. Denitrification is the conversion of NO_3^- to gaseous N products (NO, N_2O and N_2). Denitrification rate and the relative production of NO, N_2O and N_2 depend on the availability of mineral N (NH_4^+ and NO_3^-), organic C, temperature and pH, together with processes that lower the redox potential of the soil, such as changes in soil moisture. These factors not only influence the abundance of the denitrifier community, but also affect the denitrification enzyme activity in soils (Wallenstein et al. 2006).

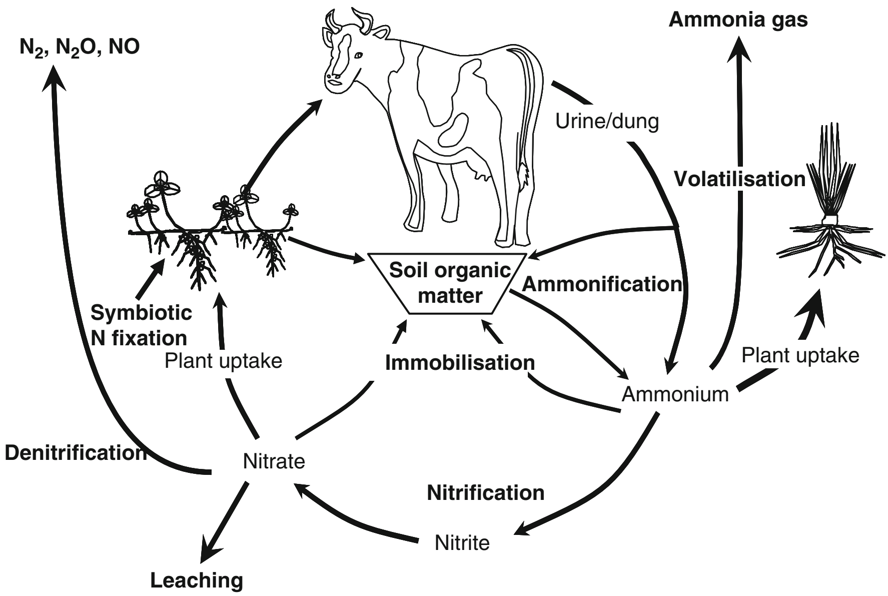

Fig. 10.2 Schematic representations of N transformations and losses in intensively managed dairy-grazed pastures (adapted from Bolan et al. 2004)

10.3.2 Nitrogen Losses

The magnitude of N input to grazed systems is generally the main factor determining the N surplus and, therefore, the potential for N losses. There are limits to how much pasture production can be increased with fertilisation. The intensively

Table 10.1 N inputs and outputs from intensive dairy farm systems in New Zealand receiving nitrogen (N) fertiliser at nil or 410 kg N ha^{-1} $year^{-1}$

	0 N	410 N
N inputs (kg N ha^{-1} $year^{-1}$)		
Clover N_2 fixation	160 (80–210)	40 (15–115)
Non-symb. fixation + atm. dep.	10	10
Fertiliser N	0	410
Purchased feed	0	41
N outputs (kg N ha^{-1} $year^{-1}$)		
Milk + meat	78 (68–83)	114 (90–135)
Transfer of excreta to lanes/sheds	53 (41–63)	77 (72–91)
Denitrification	5 (3–7)	25 (13–34)
Ammonia volatilisation	15 (15–17)	68 (47–78)
Leaching	30 (12–74)	130 (109–147)
Immobilisation of fertiliser N		70 (60–84)
N balance (kg N ha^{-1} $year^{-1}$)	−11 (−74 to +47)	17 (−11 to +24)
Farm N surplus (kg N ha^{-1} $year^{-1}$)	92	387
N use efficiency (product N/input N)	46%	23%

Bracketed values are range in N flows measured over 5 years (adapted from Ledgard et al. 2009)

managed pasture systems reach N saturation when the plants, microbes and soils cannot use or assimilate or retain more N, and additional N inputs are lost through both leaching and gaseous emissions. Ledgard et al. (1999) found that a threefold increase in total N inputs resulted in a fourfold increase in N surplus, a fourfold to fivefold increase in gaseous and leaching losses, and a halving of the N use efficiency (Table 10.1). The primary transformations leading to N losses are ammonia (NH_3) volatilisation, NO_3^- leaching and denitrification (N_2 and N_2O emissions).

10.3.2.1 Ammonia Volatilisation

In grazed pastures, biological degradation of animal excreta and hydrolysis of fertilisers containing urea and ammonium ions lead to the continuous formation of NH_3 in the soil, which can volatilise to the atmosphere. Jarvis et al. (1989) found that NH_3 loss from urine patches increased under high N fertilisation because more N was excreted in urine. Less NH_3 is lost from grazing systems than from animal housing systems, where the combined loss from the animal houses, manure storage and field application can be large. Jarvis and Ledgard (2002) compared NH_3 losses from two contrasting model dairy systems in the UK and New Zealand. Their study demonstrated distinct differences between the two farming systems in terms of total N input, N off-take, N surplus and NH_3 loss. These values were 1.7, 1.2, 1.8 and 2.4 times greater in the UK than in New Zealand, respectively. The greater loss of NH_3 in the UK farm is attributed mainly to the higher fertiliser N input, and the housing of animals and subsequent spreading of the manure on the farm. However, when NH_3 loss was expressed in relation to the farm N surplus, there was little difference between the two farms; NH_3 loss being approximately 20% of the surplus. Studies conducted in New Zealand and overseas and reviewed by Saggar et al. (2009b) have shown that fertilisers containing urea can lose up to 30% or more of their N through NH_3 emission if not rapidly incorporated into the soils. Compilation of the data using aspirated chambers from studies conducted in New Zealand by Sherlock et al. (2008) suggests the direct average NH_3-N emissions from urine applied to pasture soils are 15.9%. One method of reducing losses is to use a urease inhibitor (UI) that retards the hydrolysis of urea by soil urease and allows the urea to diffuse deeper into the soil. Much of the NH_3 then released would be retained by the soil (Saggar et al. 2009b).

10.3.2.2 Nitrate Leaching

A review of the research on grazed systems suggests that NO_3^- leaching increases exponentially with increased N inputs (Ledgard et al. 2009; Fig. 10.3).

Various studies have also shown that urine N makes a much greater contribution to NO_3^- leaching compared with fertiliser N (because of much larger specific rate of N deposition in urine). Urine typically contributes 70–90% of total N leaching loss (Monaghan et al. 2007). Fertiliser N is generally used efficiently by pastures, but it enhances pasture N uptake and grass-N concentrations, thereby increasing N excretion in urine and consequently the risk of N loss to the environment. Winter

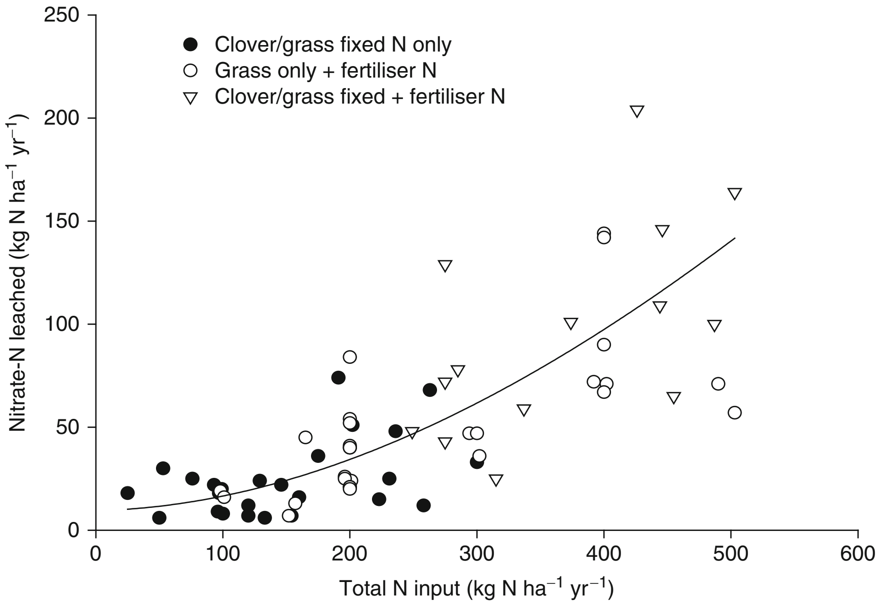

Fig. 10.3 Nitrate leaching from grazed pasture systems as affected by total N input (adapted from Ledgard et al. 2009)

leaching of N can be further exacerbated by dry summer/autumn conditions and an associated slowing down of plant growth, which results in a build-up of NO_3^- levels in soil by the end of autumn (Scholefield et al. 1993). Estimates of N leached from managed pastures vary widely, ranging from 6 to 162 kg N ha^{-1} year^{-1}, and this is due to the differences in N input, pasture N uptake, soil drainage, animal type and stocking rate (Stout et al. 2000).

Leaching of N forms other than NO_3^- is generally low. However, ammonium leaching can occur in some soils and may be enhanced where mitigation practices target reduced nitrification. Research also indicates that in some situations, dissolved organic N can be a significant source of leached N (Jones et al. 2004; Bolan et al. 2010).

Eriksen et al. (2004) observed higher leaching losses from grazed N-fertilised ryegrass pasture (on average 47 kg N ha^{-1} year^{-1}) than from grazed non-N-fertilised clover/ryegrass pasture (on average 24 kg N ha^{-1} year^{-1}). Over time the losses from the clover/ryegrass pasture decreased due to a reduction in N_2 fixation together with a reduction in dry matter production that in turn led to a lower grazing intensity and lower rate of recycling of animal excreta. The research summary of N leaching from grazed pastures in Fig. 10.3 shows overlap of N leaching values estimated from pastures with or without clover at similar N inputs. However, in long-term pastures, N inputs from N_2 fixation are usually less than 200 kg N ha^{-1} year^{-1}, thereby limiting maximum N leaching from non-N-fertilised clover/grass pastures. By contrast, N fertiliser may be used at much higher annual rates of application, with potential for high N losses.

10.3.2.3 Denitrification

The process and factors regulating denitrification are described above (see Sect. 10.3.1; Chap. 8). The annual denitrification rates in agricultural and forest soils range between 0 and 239 kg N ha^{-1} (Barton et al. 1999), with the highest rates typically occurring in irrigated, N-fertilised soils and the lowest rates occurring in native ecosystems. In New Zealand pastures, denitrification is considered the primary source of N_2O emissions as nitrification, and other aerobic transformations of urine-derived N contribute little to overall emissions (Luo et al. 1999a, b; 2008c; Saggar et al. 2004a, b, 2007b, 2009a).

10.4 Intensification Impacts on Soil Attributes

Intensively managed grazing systems can result in reductions in biodiversity, increased soil erosion and overland flow, reduced soil weight-bearing capacity, reduced soil quality and increased soil compaction. These effects are usually prominent in temperate climates under excess moisture conditions. A common concern regarding land-use intensification is the potential deterioration of soil quality or soil health. Karlen et al. (1997) defined “soil quality” as the ability of soil to function, where the soil resource is recognised as a dynamic living system comprising a balance of biological, chemical and physical processes; the two terms “soil quality” and “soil health” are considered synonymously in this book (see Chap. 1 and Preface of this book). Sparling and Schipper (2002), surveying soil quality data for 500 New Zealand soils based on land use, demonstrated a degradation in the physical properties of these soils under highly intensive land uses such as dairy farming, arable cropping and horticulture. There are many soil properties that regulate compaction and soil physical health including pore space (e.g. porosity and macroporosity), water movement (saturated and unsaturated hydraulic conductivity), resistance (penetration resistance), soil structure (aggregate size and stability) and bulk density. Soil macroporosity (or air-filled porosity) is a sensitive indicator of soil compaction (Ball et al. 2007) and soil quality. Animal treading can result in the degradation of soil physical quality through the hoof action of grazing animals (Betteridge et al. 1999, 2002; Pande et al. 2000; Ward and Greenwood 2002). These physical attributes provide the environment in which soil biological and chemical processes interact. In a recent review, MacKay (2009) identified soil erosion in hill land, compaction in low land and loss of soil organic matter in some pasture soils as additional emerging soil degradation issues of intensively managed pastoral soils.

In a review of the impacts of grazing animals on soil quality, vegetation and surface water quality in intensively managed grasslands, Bilotta et al. (2007) report that intensively managed grazing can actually lead to the degradation of both soil and vegetation by causing changes in vegetation cover and biodiversity in the pastoral sward, structural deformation of soil, changes in hydrological behaviour and deterioration of water quality within these environments. These authors quote from DEFRA (2005) and UK Environment Agency (2002) reports that ~29% of the total land area in England and Wales is intensively managed, and the damage to homes, commercial property and agricultural land from poor soil structure caused by intensification costs the UK approximately £115 million per year.

Although intensification and increased environmental damage are often associated with increased external inputs, this can also result simply from increased grazing pressures (Cuttle 2008). In terms of soil chemical and biological processes, changes in organic matter concentration, the supply and losses of nutrients, and changes in biological activity reflect the impact of intensification. Soil flora and fauna play an important role in the transformation of organic matter and in regulation of C and cycling of nutrients such as N, P and S. As farming intensifies, nutrients and chemicals are increasingly used to enhance productivity and control weeds, pests and animal diseases. For example, herbicides are used to control weeds, anti-parasitic agents are used to control gastrointestinal parasites and zinc (Zn) supplementation is used to control facial eczema in grazing animals. In addition, pathogenic organisms transferred to soil through animal excreta may transmit infection to other livestock and to humans. This section considers the impacts of intensification of pastures on key soil attributes such as soil C stocks, nutrients and physical condition.

10.4.1 Influence on Soil Carbon

Soil organic matter is important for the sustained function of agro-ecosystems as it influences chemical, biological and physical soil properties and plays a vital role in nutrient cycling (see Chap. 5). There is experimental evidence that increased utilisation of pasture biomass and increased irrigation frequency can reduce soil C content (Hoglund 1985; Lambert et al. 2000; Metherell et al. 2002). This however, is contrary to a common view that intensive pastoral agriculture can build up soil C or at best has a neutral effect. Long-term experiments in New Zealand do provide insights into the steady-state status of soil C in pastoral soils, but the soils at these sites are not without limitations. Saggar et al. (2001) found no difference in soil C levels and P status between fertilised and unfertilised soils. Process-based studies by Saggar and Hedley (2001) and Saggar et al. (1997, 2000) showed that addition of fertiliser not only increased pasture production and translocated more C to roots than non-fertilised pastures, but also enhanced the decomposition of soil organic C. Thus, increased N inputs to intensively managed pasture soils, already well supplied with N, are more likely to decrease C storage (Cuttle 2008), due to more rapid decomposition of N-enriched residues. A recent New Zealand study on soil C storage reveals that soil C has decreased on some dairy pastures but has increased on hill country pastures (Schipper et al. 2007). Investigating the relationship between the above-ground net productivity of permanent swards and soil C concentration, Bélanger et al. (1999) also found no increase in soil C concentrations from increased above-ground net productivity through N, P and K fertilisation. The loss in soil C observed with intensification mainly occurs from labile pools (Ghani et al. 2003), with implications for reduced retention of N (and other nutrients) in the soil, leading to lowered nutrient availability for plant uptake and greater losses to the environment. Soil C in sheep-grazed pastures has recently been shown to increase from the effect of increased atmospheric CO_2 concentration (Dr KR Tate pers. comm.). These results all seem to indicate the dynamic nature of soil organic matter and that several different factors, such as soil moisture, temperature, pasture growth and C input, can interact to cause changes in soil C storage. Overall, the net effect of intensification of pastoral farming on soil C can be neutral or positive or negative depending on the level of saturation of C in the soil.

10.4.2 Influence on Soil Physical Properties

As discussed, soils under pasture can accumulate soil organic matter, favouring the development of good soil structure and other properties to sustain pasture growth. However, intensification of pastoral farming can cause stress on the physical condition of pasture soils. Bilotta et al. (2007) reviewed changes in soil physical properties caused by grazing animals and showed that these can have serious implications for soil quality. They concluded that there are three main forms of soil structural change associated with grazing animals, namely compaction, pugging and poaching. In grazed pastures, animal pugging and treading damage during grazing reduce soil infiltration rates and pasture growth (Drewry et al. 2008) – a reduction that can be serious under wet soil conditions. Animal treading of pasture can also decrease soil porosity and bulk density and consequently cause an increase in mechanical impedance to root penetration and a reduction in aeration and/or an increase in water-logging of soil. This will also have a negative effect on legume growth, productivity and N_2 fixation in pasture (Menneer et al. 2004). In addition, the decrease in soil infiltration capability and hydraulic conductivity due to treading damage makes the soil more prone to ponding, and thus increases the risk of run-off losses of other nutrients, particularly P, and gaseous loss of N through denitrification from intensively grazed pastures (Monaghan et al. 2005; Bhandral et al. 2007). Changes in soil physical properties can also affect nutrient transformation processes in soil, as the changes can alter the moisture regime and influence soil respiration rates and plant nutrient uptake (Di et al. 2001). Pugging and compaction are generally more serious in areas where animals congregate, such as around paddock gateways, water troughs and in camping areas.

As discussed in this section, intensification of pastoral farming can negatively modify soil properties such as structure, permeability and soil organic matter content. These modifications change the buffering and filtering capacity of pastoral soils, for example, by increasing preferential flow and transport leading to faster and greater nutrient and contaminant leakage, favouring the potential for degradation (Ledgard and Luo 2008).

10.5 Intensification Impacts on GHG Emissions

Increased pasture production for higher per hectare animal productivity is the major goal of pastoral farmers in New Zealand, Australia, parts of South and North America, Europe, China and India. However, intensification in pastoral productivity also leads to increased emissions of the potent agricultural GHGs, CH_4 and N_2O. The global emissions of CH_4 and N_2O from grasslands-derived feeds are estimated at about 44 Tg (1 Tg = 10^{12} g = 1 million metric tones) CH_4 year^{-1} and 2.5 Tg N year^{-1}, comprising 18% and 20% of global CH_4 and N_2O emissions, respectively (Clark et al. 2005; Saggar et al. 2009a). These two non-CO_2 GHGs comprise about half of New Zealand's total emissions. Globally, N_2O production has increased by 17% from 1990 to 2005, and it has been assumed that N_2O emissions from agricultural practices will further increase by 35–60% by 2030 (IPCC 2007). Projections by Bouwman et al. (2005) estimate that in the next three decades, intensification involving improved management and use of fertilisers will be required to produce 30% more grass/animal feed to meet the global demand for meat and milk production. The impacts of future livestock intensification and fertiliser use on GHG emissions need to be assessed against the potential increases in grassland productivity and animal production.

Annual CH_4 emissions from enteric fermentation and animal manure are about 106 Tg (Steinfeld et al. 2006) globally. As livestock numbers grow, and livestock rearing becomes increasingly industrial, the production of manure is projected to increase by about 60% by 2030 resulting in similar proportional increases in enteric and manure CH_4 emissions (http://www.fao.org/docrep/004/y3557e/y3557e11.htm). Therefore, livestock CH_4 emissions are directly proportional to livestock intensification, except in situations where output per livestock unit is also increased causing reduced CH_4 emissions per unit of feed intake. Also increased intensification of grazed pastures has been shown to have a little impact on the soil CH_4 sink capacity in the Netherlands (van den Pol-Van Dasselaar et al. 1999) and New Zealand (Saggar et al. 2004c; Walcroft et al. 2008). In contrast, N_2O emissions have increased from the effects of intensification of livestock numbers, but these changes are complex and poorly understood. Increasing sheep stock numbers elevate soil N_2O emissions (e.g. Ma et al. 2006; Saggar et al. 2007a). The process-based model NZ–DNDC simulated the effects of increasing sheep stocking rates (5, 10, 15, 20 and 25 sheep ha^{-1}) and showed that soil N_2O emissions increased linearly with the stocking rates in a well-drained New Zealand pasture site (Saggar et al. 2007a) (Fig. 10.4).

This linear increase in N_2O emissions with increasing sheep numbers suggests that intensification of sheep farming may have little impact on emissions per stock unit. Saggar et al. (2007b) showed that more of the input N was used and less was

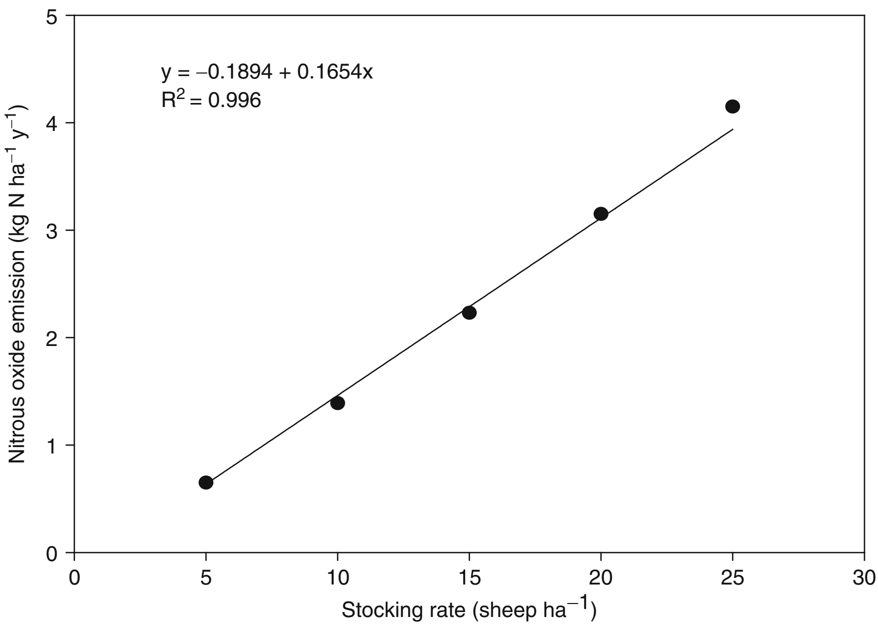

Fig. 10.4 Relationship between sheep stocking rate and nitrous oxide emissions simulated by a process-based model NZ–DNDC in a well-drained pasture site in New Zealand (data from Saggar et al. 2007a)

lost as N_2O in sheep-grazed pastures compared with dairy-grazed pastures. In steppe grassland sites in Inner Mongolia, China, a significant positive correlation was found between the stocking rate and the contribution of the growing-season emissions to the annual N_2O budget (Wolf et al. 2010). There are two main reasons for the elevated soil N_2O emissions by increasing livestock numbers. First, grazing animals excrete N in urine and dung, and N accumulates in dung and urine patches. Also synthetic N fertiliser (i.e. urea fertiliser) is often applied to enhance pasture growth for intensively managed grasslands. The excretal and synthetic N can be a source of N_2O through nitrification, denitrification and nitrifier denitrification. Second, treading and trampling by the animals cause soil compaction, making the soil more anaerobic and stimulating denitrification activity, thus facilitating N_2O production (Davidson and Firestone 1988).

It is generally assumed that there is a linear relationship between N input and direct N_2O emissions in managed agro-ecosystems (Bouwman 1996; Dobbie et al. 1999). However, there is a growing body of evidence indicating a nonlinear, exponential response of direct N_2O emissions to N input (Kim and Hernandez-Ramirez 2010). The data in this review indicated that direct N_2O emissions can increase abruptly when N input exceeds 300 kg N ha^{-1}, and there is an exponential relationship between N input and direct N_2O emissions (Fig. 10.5) and emission

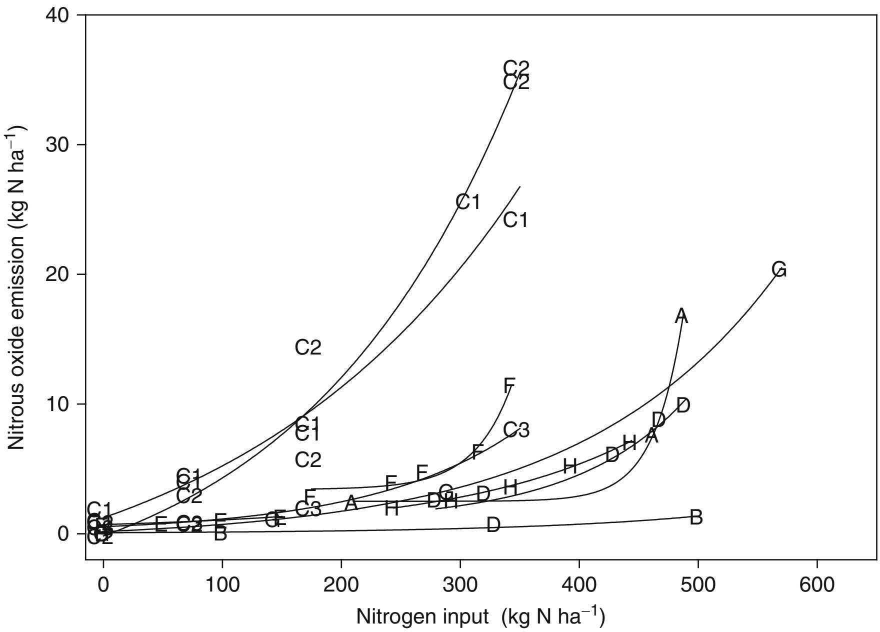

Fig. 10.5 Observed exponential relationship between N input and nitrous oxide emissions in studies conducted in grazed pasture systems. Data sources: *A* (Dobbie et al. 1999), *B* (Letica et al. 2009), *C1* (Cardenas et al. 2010; Aberystwyth site), *C2* (Cardenas et al. 2010; Aberystwyth site) and *C3* (Cardenas et al. 2010; North Yorkshire site), *D* (Hyde et al. 2006), *E* (Zhang and Han 2008), *F* (Kim et al. 2010), *G* (Singh et al. 2008) and *H* (Saggar et al. 2007a)

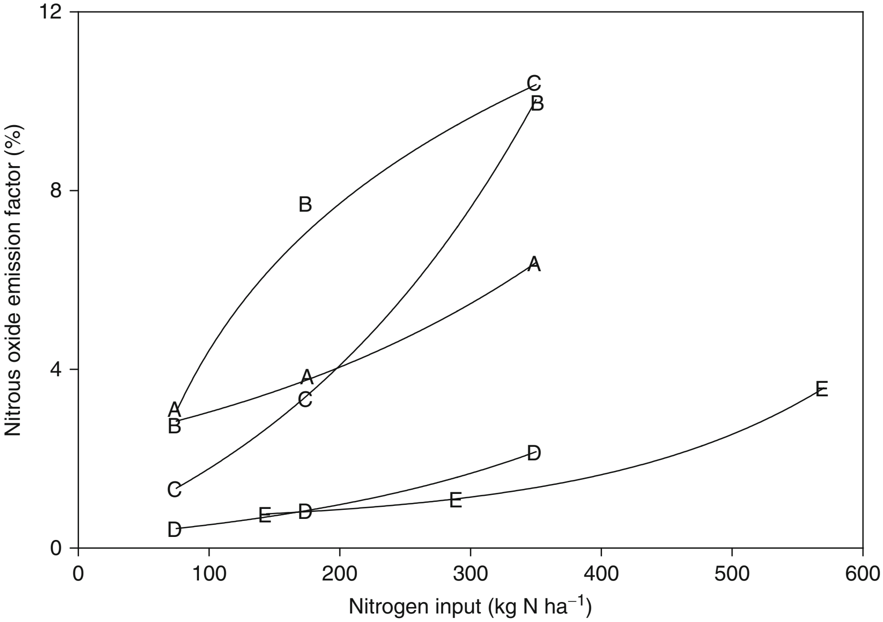

Fig. 10.6 Observed relationship between N input and nitrous oxide emission factor in studies conducted in grazed pasture systems. Data sources: *A* (Cardenas et al. 2010; Devon site), *B* (Cardenas et al. 2010; Aberystwyth site), *C3* (Cardenas et al. 2010; North Yorkshire site) and *D* (Singh et al. 2008)

factors (Fig. 10.6), attributed to excessive soil N, lower N uptake and priming effect (Kim and Hernandez-Ramirez 2010).

As the stocking density is increased, the frequency and closeness of grazing also increase. This leads to soil compaction and reduction in pore space, both of which increase N_2O emissions in grazed pastures, as found in laboratory (Uchida et al. 2008; van Groenigen et al. 2005a) and field studies (Bhandral et al. 2007; van Groenigen et al. 2005b). Repacking dairy pasture soil with four different soil aggregate sizes and four levels of soil compaction showed that the highest N_2O emissions were obtained from the smallest and most compacted aggregates (Uchida et al. 2008). Measured N_2O emissions from two pasture on well-drained and poorly drained soil grazed by dairy cows over a year following grazing events were about 2% of excretal and fertiliser N inputs (Saggar et al. 2004a), twice those determined from field-plot experiments with animal exclusion (de Klein et al. 2003). These results and those of Douglas and Crawford (1993) suggest that animal treading could accelerate N_2O emissions. Collectively, these studies (Bhandral et al. 2007; Uchida et al. 2008; van Groenigen et al. 2005a, b) show 1.3–14-fold increases in N_2O emissions with 1.1–1.4-fold increase in bulk density caused by soil compaction (Table 10.2).

Overall, intensification of pasture using high N input and stocking rate is likely to result in soil compaction, thereby causing exceptionally high gaseous and leaching losses of N. This suggests optimal N management considering stocking

Table 10.2 The effect of soil compaction on bulk density and nitrous oxide (N_2O) emission in grazed pasture systems

Site	Study type	Study periods	Soil type	N input (kg N ha^{-1})	Control (A) Bulk density (Mg m^{-3})	Control (A) N_2O emission (kg N ha^{-1})	Compacted (B) Bulk density (Mg m^{-3})	Compacted (B) N_2O emission (kg N ha^{-1})	Difference (B/A) Bulk density	Difference (B/A) N_2O emission	References
Palmerston North, New Zealand	Field	10 September to 4 December 2002	Fine sandy loam	Urine, 600	1.18	2.9	1.31	9.2	1.1	3.1	Bhandral et al. (2007)
				Nitrate, 600		4.4		61.5		14.1	
				Ammonium, 600		2.6		9.2		3.5	
				Urea, 600		2.1		9.1		4.3	
				Water, 0		1.12		2.61		2.3	
Lincoln, New Zealand	Laboratory	37 days	Silt loam	Urine, 340	0.97	13.3	1.29	32.6	1.3	2.5	Uchida et al. (2008)
					0.78	4.1	1.08	30.3	1.4	7.4	
					0.82	1.0	1.13	8.5	1.4	8.3	
					0.81	1.0	1.12	1.4	1.4	1.3	
Wageningen, The Netherlands	Laboratory	103 days	Sandy	Urine, 119[a]	1.07	1.0[a]	1.22	8.2[a]	1.1	8.2	van Groenigen et al. (2005a)
				Urine, 237[a]	1.07	1.7[a]	1.22	13.0[a]	1.1	7.6	
				Urine, 474[a]	1.07	5.4[a]	1.22	27.2[a]	1.1	5.0	
				Urine, 949[a]	1.07	9.5[a]	1.22	16.1[a]	1.1	1.7	
Wageningen, The Netherlands	Field	August 2000 to November 2001	Sandy	Urine, 186	1.54	1.7	1.6	6.2	1.04	3.6	van Groenigen et al. (2005b)
				Urine, 373		6.2		9.4		1.5	

[a]Unit: mg N kg^{-1}

rate and expected pasture yields is the key to mitigating N losses in grazed pasture systems.

10.6 Approaches to Reduce Intensification Impacts

Meat and milk products are considered global public goods because of their role in the world food supply. Concerns about food safety and security, energy security, biosecurity and traceability are gaining significance as consumers recognise the relationship between diet and health. In the last decade, there has been a rapid rise in the combined consumption of meat and milk globally, and there is an expectation that the total demand for livestock products might almost double by 2050 (Herrero et al. 2009). This increasing demand for animal products has major economic and environmental implications for countries whose economies are based primarily on livestock farming. In a carbon-constrained post-peak oil era, these countries will need to have globally competitive and sustainable livestock farming systems in place, and have strategies developed and ready for implementation to manage the risks and opportunities from global climate change. These strategies will also need to meet stringent goals for sustainability, environmental security, economy and society. They will also have to be able to adapt in response to changing circumstances. Therefore, sustainable livestock production goals will need to balance livestock production, livelihoods and environmental protection (Herrero et al. 2009).

A range of practices and technologies has been examined in New Zealand to mitigate adverse environmental effects due to intensification (de Klein and Eckard 2008; Di and Cameron 2006; Luo et al. 2008a, b, d, 2010; Saggar et al. 2009a). These practices and technologies include soil management (Uchida et al. 2008; Velthof et al. 2009; Zaman et al. 2008), the use of winter stand-off/feed pads or housing systems during high-risk periods of N and other nutrient loss (Ledgard et al. 2006; Luo et al. 2006), integration of low protein or condensed tannin forages (Luo et al. 2008a; Nielsen et al. 2003), improved management of N fertilisers (Luo et al. 2007) and the use of nitrification inhibitors (NIs) (Asing et al. 2008; Di and Cameron 2006; Singh et al. 2008; Zaman et al. 2009).

10.6.1 Soil Management

A number of studies have shown that reducing N_2O and NO_3^- losses and soil CH_4 production could be achieved by altering soil conditions including application of lime and zeolite (Zaman et al. 2007, 2008) and biochar (Spokas et al. 2009; Yanai et al. 2007), improving drainage (de Klein et al. 2003), and avoiding soil compaction (Livesley et al. 2008; Uchida et al. 2008; van Groenigen et al. 2005a, b). Among these, biochar application has attracted more interest recently, as it has been hypothesised that this can achieve C sequestration and may reduce net GHG emissions (Lehmann and Joseph 2009; Sohi et al. 2010). However, research so far is very limited, with conflicting results (Chap. 15). Biochar incorporation in soil has reduced GHG emissions in laboratory experiments conducted in Japan (Yanai et al. 2007), the USA (Spokas et al. 2009), and Australia (Singh et al. 2010). However, a recent New Zealand laboratory study (Clough et al. 2010) showed that biochar application did not affect N_2O emissions but enhanced NH_3 emissions. Further studies are clearly needed to evaluate the effect of biochar addition on GHG emissions where factors such as biochar type and soil properties are studied (Chap. 15). Renewal of grazed pastures can cause high N_2O emissions (e.g. Davies et al. 2001; Mori and Hojito 2007) with higher emissions after renewal without ploughing than with ploughing (Velthof et al. 2009). There are conflicting results on the appropriate season for mitigating N_2O emissions caused by renovation. While it is expected that pasture renovation in spring instead of autumn might offer opportunities to lower N_2O emission (Vellinga et al. 2004), Velthof et al. (2009) found higher N_2O emissions after renovation in spring than in autumn.

10.6.2 Winter Stand-Off/Feed Pads or Housing Systems

Soil compaction can be minimised through farm management practices, including reduced stocking rates and length of grazing rotation, avoiding grazing in wet soil conditions, and improving soil drainage (Greenwood and McKenzie 2001; Singleton and Addison 1999). About 7% of New Zealand N_2O emission can be reduced by following management practice avoiding soil compaction (de Klein and Ledgard 2005).

Stand-off/feed pads or housing systems have been used to reduce soil physical damage due to grazing on wet soils. They can also reduce N_2O emissions and NO_3^- leaching (Ledgard et al. 2006; Luo et al. 2006) because of lower excreta input to the soil and less soil compaction during the wet winter and early spring seasons. In a limited number of studies in New Zealand, N_2O emissions and N leaching were reduced by up to 60% when animals were held on stand-off/feed pads or in animal houses for 3–4 months during late-autumn–winter periods compared with year-round grazing (Chadwick et al. 2002; de Klein et al. 2006; Ledgard et al. 2006; Luo et al. 2008b). Use of a stand-off pad decreased total N_2O emissions per hectare of a dairy farmlet by 9%, compared with the control farm (Luo et al. 2010).

10.6.3 Integration of Low Protein or Condensed Tannin Forages

A lower proportion of N was excreted in urine and faeces when animals grazing perennial ryegrass pasture were fed supplements containing a low protein concentration and highly fermentable organic matter (Mulligan et al. 2004; Nielsen et al. 2003). Results from a study by Luo et al. (2008a) suggest that integration of low protein forage can be an effective management practice to mitigate adverse environmental effects such as N_2O emission intensity with higher stocking rates in dairy farm systems,

10.6.4 Management of N Fertilisers

Since NO_3^-, leaching and N_2O emissions following fertiliser application can be elevated in wet soils (Luo et al. 2007), strategic application of N fertilisers and farm dairy effluent to pastures under low soil moisture status can potentially reduce N losses (Luo et al. 2008b). Limiting the amount of N fertiliser applied during late-autumn/winter or early spring, when pasture growth is slow and soil is wet, can decrease N losses from grazed pastures.

10.6.5 Use of Nitrogen Transformation Inhibitors

NIs such as dicyandiamide (DCD), nitrapyrin and 3,4 dimethylpyrazole phosphate (DMPP) slow the activity of nitrifying bacteria responsible for the oxidation of NH_4^+ to NO_2^- and can thereby reduce NO_3^- leaching and N_2O emissions (Abbsi and Adams 2000; Cameron et al. 2005; Di et al. 2007). Ammonia emissions can be reduced using UIs such as [*N*-(*n*-butyl) thiophosphoric acid triamide; nBTPT] sold under the trade name Agrotain®, which reduce the rate of urea hydrolysis to NH_4^+ (Saggar et al. 2009b). Both NO_3^- leaching and N_2O emissions from urine patches can be potentially reduced by up to 70% with land application of NI to pastures (Asing et al. 2008; Di and Cameron 2006; Zaman et al. 2009). In addition, N is held in the NH_4^+ form longer, encouraging NH_4^+ uptake by pasture plants and preventing N_2O emissions from either nitrification or denitrification. However, this may also increase NH_3 emissions and potential NH_4^+-N leaching from urea fertiliser and cattle urine (Singh 2007). Results of New Zealand studies reviewed by Saggar et al. (2009b) suggest UI Agrotain reduced NH_3 emissions on average by 43% from urea and by 48% from urine. Some more recent studies have found that the combined use of NI (DCD) and (UI) (nBTPT) can be very effective in reducing NH_3 and N_2O emissions and NO_3^- leaching (Singh et al. 2008; Zaman and Blennerhassett 2010; Zaman et al. 2009).

Overall, the key mitigation options for reducing gaseous and leaching losses of N from intensive pastoral farming are: (1) N transformation inhibitors (NI and UI), (2) strategic farm effluent irrigation and (3) restricted winter grazing.

10.6.6 Enhancement of Soil Uptake of CH_4

There are as yet no cost-effective technologies and strategies available to livestock farmers to reduce enteric CH_4 emissions. As soils contain both CH_4-producing (methanogens) and CH_4-oxidising (methanotrophs: soil bacteria that use CH_4 as a sole C source) organisms, they have the ability to produce and consume CH_4 simultaneously (see Chap. 8). Globally, these methanotrophs remove about 10–44 Tg of CH_4 from the atmosphere. However, the methanotrophs are more important than this figure might indicate, as they also consume a great deal of CH_4 before it is released to the atmosphere.

Saggar et al. (2008) indicated that the key questions that need to be addressed for enhancing soil CH_4 uptake rates in the field are:

- What are the key microbiological populations and processes responsible for CH_4 oxidation in soils and can they be optimised?
- How do soil/plant/animal interactions and climate affect net CH_4 oxidation rates and the microbial populations regulating them, and what are the opportunities for enhancing oxidation rates?

A novel approach for capturing CH_4 produced by animals and animal effluent in confined locations (e.g. waste ponds and barns) being investigated is use of methanotrophs (Pratt et al. 2010). Biofiltration, using very active methanotroph populations in porous media to convert CH_4 to CO_2, appears to be a potentially effective strategy for treating CH_4 emissions from waste ponds on dairy farms. Melse and van der Werf (2005) observed an 85% removal efficiency of CH_4 emissions from piggery effluent using biofiltration. As the methanotrophs can rapidly consume atmospheric CH_4, they offer the potential to capture the enteric CH_4 from housed animals, effluent ponds and also emissions from landfills using biofilters to convert this potent gas to CO_2.

10.6.7 Carbon Sequestration

Soil C could be sequestered in grazed pasture systems using a range of management practices (Chan et al. 2010; Conant and Paustian 2002; Herrero et al. 2009; Reid et al. 2004). Conant and Paustian (2002) found that universal rehabilitation of overgrazed grasslands can sequester approximately 45 Tg C year^{-1} by cessation of overgrazing and implementation of moderate grazing intensity. It was suggested that soil C sequestration can be achieved by traditional pastoral practices and knowledge, such as managing grazing intensity and duration, improving pasture quality, reducing the frequency and/or intensity of grassland fires, and by providing pastoral farmers with food security benefits at the same time (Reid et al. 2004) and by management practices aimed at increasing N retention at the landscape level (Pineiro et al. 2010).

It is certainly true that C cycles rapidly through pastoral systems, and that farming ruminant animals do not add any "new" C to the atmosphere. However, in the process of rumination, some of the C in the atmosphere is transformed from a gas with a lower GWP (CO_2) to a gas with a higher GWP (CH_4). Some livestock farmers believe that increasing pasture production will lead to more C stored in the soil. Parson et al. (2009) report that increasing stocking rate in general should decrease the flow of C to soil, and so reduce the potential to increase soil C. However, the factors (such as higher soil fertility) that increase plant growth increase overall C flows to soil and may increase soil C. In New Zealand, the potential for significant, permanent increases in soil organic C in intensive pasture systems is limited (Whitehead et al. 2009).

10.6.8 Farm Economics

Reduced agricultural intensity can decrease N emissions and other inverse environmental effects, but would have a major impact on economic returns and farm viability. Productivity and environmental gains occur through above-mentioned practices and technologies by avoiding losses of N and other nutrients to the environment and increasing the quantity and quality of the forage produced. A recent modelling study of the production, environmental and financial impacts of intensification of New Zealand sheep and beef farming systems found increases in total N leaching and GHG emissions from intensification through both feeding maize silage and applying N fertilisers (White et al. 2010). These model estimates also show that neither method of intensification increased profitability without a small annual N application of 50 kg N ha^{-1}, especially to 75% of hill country farms. Other New Zealand studies discussed in this chapter clearly show the advantages of instigating several of these individual dairy farm management practices such as the use of winter stand-off pads, maize silage and improved N fertiliser management, in reducing N_2O emissions and N leaching (Luo et al. 2010). Results of a 3-year CO_2, N_2O and CH_4 emission measurements in an upland semi-natural grassland site grazed by cattle (Allard et al. 2007) showed that reducing fertiliser input and grazing pressure strongly reduced N_2O and CH_4 emissions per unit of land area but gradually reduced the C storage potential of the grassland. These results clearly demonstrate the need for taking into account all the three major GHGs (CO_2, N_2O and CH_4) when developing strategies to mitigate the GHG effect.

10.7 Conclusions and Future Work

Intensification of pastoral agriculture has occurred since the establishment of managed pastoral systems and will continue to occur in future to meet growing demands for food worldwide. The key drivers of intensification are the need to maintain or increase profit and return on investment, containing cost of inputs relative to the value of outputs, the availability of new knowledge and technologies, high land values and increasing international competition, and possible tariff barriers for unsustainable practices.

Intensively managed livestock farming changes the buffering and filtering capacity of our structured pastoral soils and lead to faster and greater nutrient and contaminant leakage, favouring the potential for degradation, increased nutrient losses to waterways and the atmosphere, and increased environmental pollution. Intensification of pastoral productivity also leads to increased emissions of the principal agricultural GHGs, CH_4 and N_2O. Some of the contaminants have the potential to disrupt wildlife welfare and human health. With the current pressures exerted on grassland resources, it is not possible to continue to increase productivity without causing further soil, vegetation and environmental degradation.

Concerns about food safety and security, energy security, biosecurity and traceability are gaining significance as consumers recognise the relationship between their diet and health. Agricultural policies based on sound science and robust risk assessments are needed, and research efforts must be directed towards achieving a balance between environmentally sustainable management of pastoral resources and efficient food production. Further studies are needed to quantify organic matter dynamics and consequent nutrient fluxes in soils, and associated soil-atmosphere exchange of GHGs under different livestock-based land uses. These studies need to be coupled with predicting water and solute storage and movement through soils and land systems. The information is needed to assess and/or mitigate the nutrient losses for robust models that combine farm systems expertise and provide information for developing solutions at multiple scales. Efforts to integrate remote-sensing and geographic information system capabilities need to be expanded from on-farm management systems to provide information for spatial modelling and forecasting at multiple scales. Simultaneously, strong linkages with landowners, community groups and regional authorities are needed to unify and use aspirations through sustainable practices.

Researchers and policy makers need to consider the whole food chain, and to account for multiple environmental emissions and resource efficiency, for example, as energy use under intensive livestock farming, through the use of tools such as life cycle assessment. This type of assessment can help identify potential (and unintended) issues such as pollution swapping or likely additional energy costs associated with a particular management system, processing and mitigation option. These approaches are essential not only to protect food-producing countries, but also to avoid the imposition of tariff barriers in distant markets.

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