03

Livestock's role in climate change and air pollution

3.1 Issues and trends

The atmosphere is fundamental to life on earth. Besides providing the air we breathe it regulates temperature, distributes water, it is a part of key processes such as the carbon, nitrogen and oxygen cycles, and it protects life from harmful radiation. These functions are orchestrated, in a fragile dynamic equilibrium, by a complex physics and chemistry. There is increasing evidence that human activity is altering the mechanisms of the atmosphere.

In the following sections, we will focus on the anthropogenic processes of climate change and air pollution and the role of livestock in those processes (excluding the ozone hole). The contribution of the livestock sector as a whole to these processes is not well known. At virtually each step of the livestock production process substances contributing to climate change or air pollution, are emitted into the atmosphere, or their sequestration in other reservoirs is hampered. Such changes are either the direct effect of livestock rearing, or indirect contributions from other steps on the long road that ends with the marketed animal product. We will analyse the most important processes in their order in the food chain, concluding with an assessment of their cumulative effect. Subsequently a number of options are presented for mitigating the impacts.

Climate change: trends and prospects

Anthropogenic climate change has recently become a well established fact and the resulting impact on the environment is already being observed. The greenhouse effect is a key mechanism of temperature regulation. Without it, the average temperature of the earth's surface would not be 15ºC but -6ºC. The earth returns energy received from the sun back to space by reflection of light and by emission of heat. A part of the heat flow is absorbed by so-called greenhouse gases, trapping it in the atmosphere. The principal greenhouse gases involved in this process include carbon dioxide (CO_2), methane (CH_4) nitrous oxide (N_2O) and chlorofluorocarbons. Since the beginning of the industrial period anthropogenic emissions have led to an increase in concentrations of these gases in the atmosphere, resulting in global warming. The average temperature of the earth's surface has risen by 0.6 degrees Celsius since the late 1800s.

Recent projections suggest that average temperature could increase by another 1.4 to 5.8 °C by 2100 (UNFCCC, 2005). Even under the most optimistic scenario, the increase in average temperatures will be larger than any century-long trend in the last 10 000 years of the present-day interglacial period. Ice-core-based climate records allow comparison of the current situation with that of preceding interglacial periods. The Antarctic Vostok ice core, encapsulating the last 420 000 years of Earth history, shows an overall remarkable correlation between greenhouse gases and climate over the four glacial-interglacial cycles (naturally recurring at intervals of approximately 100 000 years). These findings were recently confirmed by the Antarctic Dome C ice core, the deepest ever drilled, representing some 740 000 years - the longest, continuous, annual climate record extracted from the ice (EPICA, 2004). This confirms that periods of CO_2 build-up have most likely contributed to the major global warming transitions at the earth's surface. The results also show that human activities have resulted in

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Cracked clay soil – Tunisia 1970

present-day concentrations of CO_2 and CH_4 that are unprecedented over the last 650 000 years of earth history (Siegenthaler *et al.*, 2005).

Global warming is expected to result in changes in weather patterns, including an increase in global precipitation and changes in the severity or frequency of extreme events such as severe storms, floods and droughts.

Climate change is likely to have a significant impact on the environment. In general, the faster the changes, the greater will be the risk of damage exceeding our ability to cope with the consequences. Mean sea level is expected to rise by 9–88 cm by 2100, causing flooding of low-lying areas and other damage. Climatic zones could shift poleward and uphill, disrupting forests, deserts, rangelands and other unmanaged ecosystems. As a result, many ecosystems will decline or become fragmented and individual species could become extinct (IPCC, 2001a).

The levels and impacts of these changes will vary considerably by region. Societies will face new risks and pressures. Food security is unlikely to be threatened at the global level, but some regions are likely to suffer yield declines of major crops and some may experience food shortages and hunger. Water resources will be affected as precipitation and evaporation patterns change around the world. Physical infrastructure will be damaged, particularly by the rise in sea-level and extreme weather events. Economic activi-

Box 3.1 The Kyoto Protocol

In 1995 the UNFCCC member countries began negotiations on a protocol – an international agreement linked to the existing treaty. The text of the so-called Kyoto Protocol was adopted unanimously in 1997; it entered into force on 16 February 2005.

The Protocol's major feature is that it has mandatory targets on greenhouse-gas emissions for those of the world's leading economies that have accepted it. These targets range from 8 percent below to 10 percent above the countries' individual 1990 emissions levels "with a view to reducing their overall emissions of such gases by at least 5 percent below existing 1990 levels in the commitment period 2008 to 2012". In almost all cases – even those set at 10 percent above 1990 levels – the limits call for significant reductions in currently projected emissions.

To compensate for the sting of these binding targets, the agreement offers flexibility in how countries may meet their targets. For example, they may partially compensate for their industrial, energy and other emissions by increasing "sinks" such as forests, which remove carbon dioxide from the atmosphere, either on their own territories or in other countries.

Or they may pay for foreign projects that result in greenhouse-gas cuts. Several mechanisms have been established for the purpose of emissions trading. The Protocol allows countries that have unused emissions units to sell their excess capacity to countries that are over their targets. This so-called "carbon market" is both flexible and realistic. Countries not meeting their commitments will be able to "buy" compliance but the price may be steep. Trades and sales will deal not only with direct greenhouse gas emissions. Countries will get credit for reducing greenhouse gas totals by planting or expanding forests ("removal units") and for carrying out "joint implementation projects" with other developed countries – paying for projects that reduce emissions in other industrialized countries. Credits earned this way may be bought and sold in the emissions market or "banked" for future use.

The Protocol also makes provision for a "clean development mechanism," which allows industrialized countries to pay for projects in poorer nations to cut or avoid emissions. They are then awarded credits that can be applied to meeting their own emissions targets. The recipient countries benefit from free infusions of advanced technology that for example allow their factories or electrical generating plants to operate more efficiently – and hence at lower costs and higher profits. The atmosphere benefits because future emissions are lower than they would have been otherwise.

Source: UNFCCC (2005).

ties, human settlements, and human health will experience many direct and indirect effects. The poor and disadvantaged, and more generally the less advanced countries are the most vulnerable to the negative consequences of climate change because of their weak capacity to develop coping mechanisms.

Global agriculture will face many challenges over the coming decades and climate change will complicate these. A warming of more than 2.5°C could reduce global food supplies and contribute to higher food prices. The impact on crop yields and productivity will vary considerably. Some agricultural regions, especially in the tropics and subtropics, will be threatened by climate change, while others, mainly in temperate or higher latitudes, may benefit.

The livestock sector will also be affected. Livestock products would become costlier if agricultural disruption leads to higher grain prices. In general, intensively managed livestock systems will be easier to adapt to climate change than will crop systems. Pastoral systems may not adapt so readily. Pastoral communities tend to adopt new methods and technologies more slowly, and livestock depend on the productivity and quality of rangelands, some of which may be adversely affected by climate change. In addition, extensive livestock systems are more susceptible to changes in the severity and distribution of livestock diseases and parasites, which may result from global warming.

As the human origin of the greenhouse effect became clear, and the gas emitting factors were identified, international mechanisms were created to help understand and address the issue. The United Nations Framework Convention on Climate Change (UNFCCC) started a process of international negotiations in 1992 to specifically address the greenhouse effect. Its objective is to stabilize greenhouse gas concentrations in the atmosphere within an ecologically and economically acceptable timeframe. It also encourages research and monitoring of other possible environmental impacts, and of atmospheric chemistry. Through its legally binding Kyoto Protocol, the UNFCCC focuses on the direct warming impact of the main anthropogenic emissions (see Box 3.1). This chapter concentrates on describing the contribution of livestock production to these emissions. Concurrently it provides a critical assessment of mitigation strategies such as emissions reduction measures related to changes in livestock farming practices.

The direct warming impact is highest for carbon dioxide simply because its concentration and the emitted quantities are much higher than that of the other gases. Methane is the second most important greenhouse gas. Once emitted, methane remains in the atmosphere for approximately 9–15 years. Methane is about 21 times more effective in trapping heat in the atmosphere than carbon dioxide over a 100-year period. Atmospheric concentrations of CH_4 have increased by about 150 percent since pre-industrial times (Table 3.1), although the rate of increase has been declining recently. It is emitted from a variety of natural and human-influenced sources. The latter include landfills, natural gas and petroleum systems, agricultural activities, coal mining, stationary and mobile combustion, wastewater treatment and certain industrial process (US-EPA, 2005). The IPCC has estimated that slightly more than half of the current CH_4 flux to the atmosphere is anthropogenic (IPCC, 2001b). Total global anthropogenic CH_4 is estimated to be 320 million tonnes CH_4/yr, i.e. 240 million tonnes of carbon per year (van Aardenne *et al.*, 2001). This total is comparable to the total from natural sources (Olivier *et al.*, 2002).

Nitrous oxide, a third greenhouse gas with important direct warming potential, is present in the atmosphere in extremely small amounts. However, it is 296 times more effective than carbon dioxide in trapping heat and has a very long atmospheric lifetime (114 years).

Livestock activities emit considerable amounts of these three gases. Direct emissions from livestock come from the respiratory process of all animals in the form of carbon dioxide. Ruminants, and to a minor extent also monogastrics,

Table 3.1

Past and current concentration of important greenhouse gases

Gas	Pre-industrial concentration (1 750)	Current tropospheric concentration	Global warming potential*
Carbon dioxide (CO_2)	277 ppm	382 ppm	1
Methane (CH_4)	600 ppb	1 728 ppb	23
Nitrous oxide (N_2O)	270–290 ppb	318 ppb	296

Note: ppm = parts per million; ppb = parts per billion; ppt = parts per trillion; *Direct global warming potential (GWP) relative to CO_2 for a 100 year time horizon. GWPs are a simple way to compare the potency of various greenhouse gases. The GWP of a gas depends not only on the capacity to absorb and reemit radiation but also on how long the effect lasts. Gas molecules gradually dissociate or react with other atmospheric compounds to form new molecules with different radiative properties.
Source: WRI (2005); 2005 CO_2: NOAA (2006); GWPs: IPCC (2001b).

emit methane as part of their digestive process, which involves microbial fermentation of fibrous feeds. Animal manure also emits gases such as methane, nitrous oxides, ammonia and carbon dioxide, depending on the way they are produced (solid, liquid) and managed (collection, storage, spreading).

Livestock also affect the carbon balance of land used for pasture or feedcrops, and thus indirectly contribute to releasing large amounts of carbon into the atmosphere. The same happens when forest is cleared for pastures. In addition, greenhouse gases are emitted from fossil fuel used in the production process, from feed production to processing and marketing of livestock products. Some of the indirect effects are difficult to estimate, as land use related emissions vary widely, depending on biophysical factors as soil, vegetation and climate as well as on human practices.

Air pollution: acidification and nitrogen deposition

Industrial and agricultural activities lead to the emission of many other substances into the atmosphere, many of which degrade the quality of the air for all terrestrial life.[1] Important examples of air pollutants are carbon monoxide, chlorofluorocarbons, ammonia, nitrogen oxides, sulphur dioxide and volatile organic compounds.

In the presence of atmospheric moisture and oxidants, sulphur dioxide and oxides of nitrogen are converted to sulphuric and nitric acids. These airborne acids are noxious to respiratory systems and attack some materials. These air pollutants return to earth in the form of acid rain and snow, and as dry deposited gases and particles, which may damage crops and forests and make lakes and streams unsuitable for fish and other plant and animal life. Though usually more limited in its reach than climate change, air pollutants carried by winds can affect places far (hundreds of kilometres if not further) from the points where they are released.

The stinging smell that sometimes stretches over entire landscapes around livestock facilities is partly due to ammonia emission.[2] Ammonia volatilization (nitrified in the soil after deposition) is among the most important causes of acidifying wet and dry atmospheric deposition, and a large part of it originates from livestock excreta. Nitrogen (N) deposition is higher in northern Europe than elsewhere (Vitousek *et al.*, 1997). Low-level increases in nitrogen deposition associated with air pollution have been implicated in forest productivity increases over large regions. Temperate and boreal forests, which historically have been nitrogen-limited, appear to be most affected. In areas that become nitrogen-saturated, other nutrients are leached from the soil, resulting eventually in forest dieback – counteracting, or even overwhelming, any growth-enhancing effects of CO_2 enrichment. Research shows that in 7–18 percent of the global area of (semi-) natural ecosystems, N deposition substantially exceeds the critical load, presenting a risk of eutrophication and increased leaching (Bouwman and van Vuuren, 1999) and although knowledge of the impacts of N deposition at the global level is still limited, many biologically valuable areas may be affected (Phoenix *et al.*, 2006). The risk is particularly high in Western Europe, in large parts of which over 90 percent of the vulnerable ecosystems receive more than the critical load of nitrogen. Eastern Europe and North America are subject to medium risk levels. The results suggest that even a number of regions with low population densities, such as Africa and South America, remote regions of Canada and the Russian Federation, may become affected by N eutrophication.

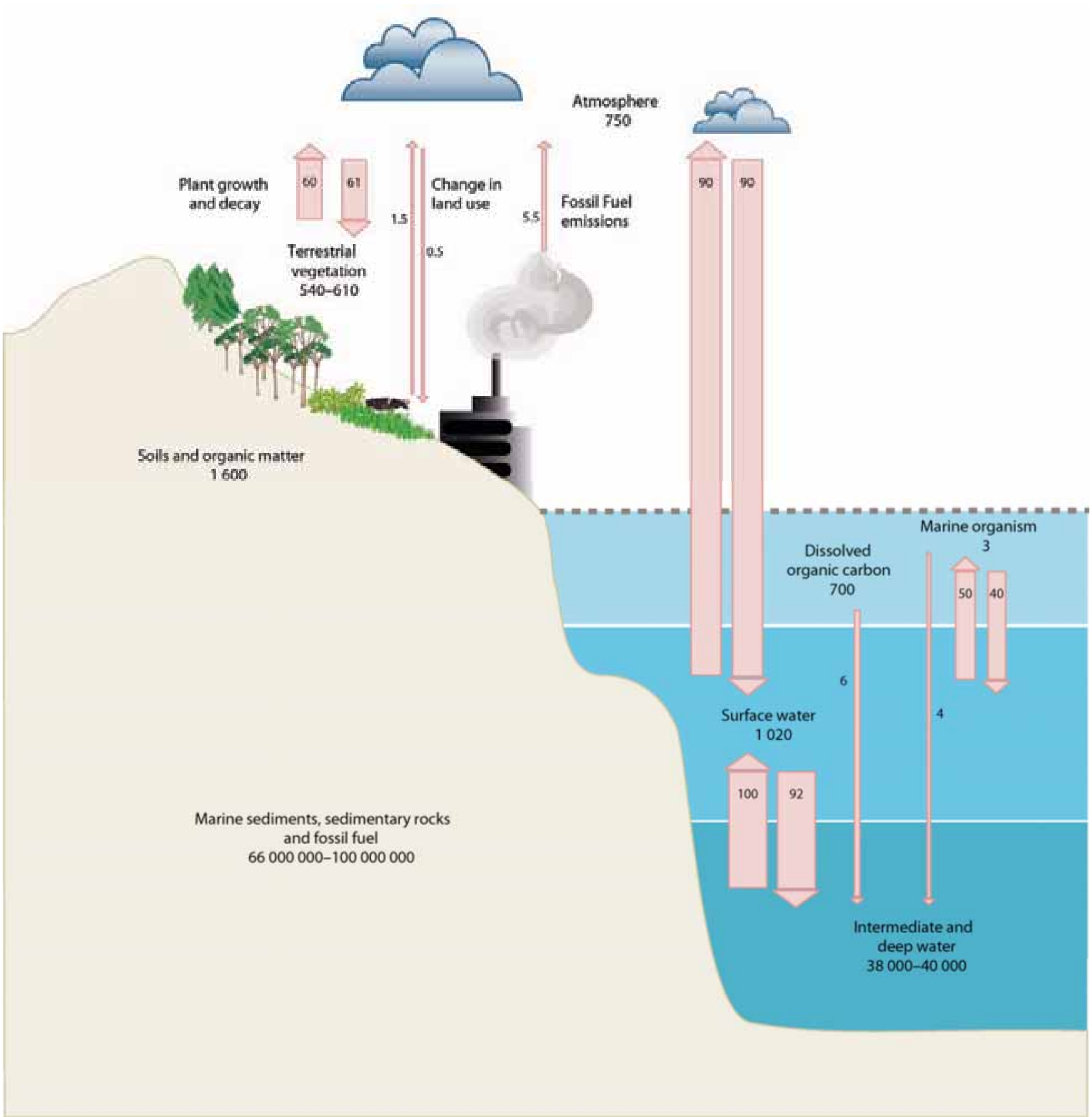
[1] The addition of substances to the atmosphere that result in direct damage to the environment, human health and quality of life is termed air pollution.

[2] Other important odour-producing livestock emissions are volatile organic compounds and hydrogen sulphide. In fact, well over a hundred gases pass into the surroundings of livestock operations (Burton and Turner, 2003; NRC, 2003).

3.2 Livestock in the carbon cycle

The element carbon (C) is the basis for all life. It is stored in the major sinks shown in Figure 3.1 which also shows the relative importance of the main fluxes. The global carbon cycle can be divided into two categories: the geological, which operates over large time scales (millions of years), and the biological/physical, which operates at shorter time scales (days to thousands of years).

Figure 3.1 **The present carbon cycle**



Note: Volumes and exchanges in billion tonnes of carbon. The figures present annual averages over the period 1980 to 1989. The component cycles are simplified. Evidence is accumulating that many of the fluxes can significantly change from year to year. Although this figure conveys a static view, in the real world the carbon system is dynamic and coupled to the climate system on seasonal, interannual and decadal timescales.
Source: Adapted from UNEP-GRID Vital Climate Graphics (available at www.grida.no/climate/vital/13.htm).

Ecosystems gain most of their carbon dioxide from the atmosphere. A number of autotrophic organisms[3] such as plants have specialized mechanisms that allow for absorption of this gas into their cells. Some of the carbon in organic matter produced in plants is passed to the heterotrophic animals that eat them, which then exhale it into the atmosphere in the form of carbon dioxide. The CO_2 passes from there into the ocean by simple diffusion.

Carbon is released from ecosystems as carbon dioxide and methane by the process of respiration that takes place in both plants and animals. Together, respiration and decomposition (respiration mostly by bacteria and fungi that consumes organic matter) return the biologically fixed carbon back to the atmosphere. The amount of carbon taken up by photosynthesis and released back to the atmosphere by respiration each year is 1 000 times greater than the amount of carbon that moves through the geological cycle on an annual basis.

Photosynthesis and respiration also play an important role in the long-term geological cycling of carbon. The presence of land vegetation enhances the weathering of rock, leading to the long-term—but slow—uptake of carbon dioxide from the atmosphere. In the oceans, some of the carbon taken up by phytoplankton settles to the bottom to form sediments. During geological periods when photosynthesis exceeded respiration, organic matter slowly built up over millions of years to form coal and oil deposits. The amounts of carbon that move from the atmosphere, through photosynthesis and respiration, back to the atmosphere are large and produce oscillations in atmospheric carbon dioxide concentrations. Over the course of a year, these biological fluxes of carbon are over ten times greater than the amount of carbon released to the atmosphere by fossil fuel burning. But the anthropogenic flows are one-way only, and this characteristic is what leads to imbalance in the global carbon budget. Such emissions are either net additions to the biological cycle, or they result from modifications of fluxes within the cycle.

[3] Autotrophic organisms are auto-sufficient in energy supply, as distinguished from parasitic and saprophytic; heterotrophic organisms require an external supply of energy contained in complex organic compounds to maintain their existence.

Livestock's contribution to the net release of carbon

Table 3.2 gives an overview of the various carbon sources and sinks. Human populations, economic growth, technology and primary energy requirements are the main driving forces of anthropogenic carbon dioxide emissions (IPCC – special report on emission scenarios).

The net additions of carbon to the atmosphere are estimated at between 4.5 and 6.5 billion tonnes per year. Mostly, the burning of fossil fuel and land-use changes, which destroy organic carbon in the soil, are responsible.

The respiration of livestock makes up only a very small part of the net release of carbon that

Table 3.2

Atmospheric carbon sources and sinks

Factor	Carbon flux (billion tonnes C per year)	
	Into the atmosphere	Out of the atmosphere
Fossil fuel burning	4–5	
Soil organic matter oxidation/erosion	61–62	
Respiration from organisms in biosphere	50	
Deforestation	2	
Incorporation into biosphere through photosynthesis		110
Diffusion into oceans		2.5
Net	117–119	112.5
Overall annual net increase in atmospheric carbon	+4.5–6.5	

Source: available at www.oznet.ksu.edu/ctec/Outreach/science_ed2.htm can be attributed to the livestock sector. Much more is released indirectly by other channels including:

- burning fossil fuel to produce mineral fertilizers used in feed production;
- methane release from the breakdown of fertilizers and from animal manure;
- land-use changes for feed production and for grazing;
- land degradation;
- fossil fuel use during feed and animal production; and
- fossil fuel use in production and transport of processed and refrigerated animal products.

In the sections that follow we shall look at these various channels, looking at the various stages of livestock production.

3.2.1 Carbon emissions from feed production

Fossil fuel use in manufacturing fertilizer may emit 41 million tonnes of CO_2 per year

Nitrogen is essential to plant and animal life. Only a limited number of processes, such as lightning or fixation by rhizobia, can convert it into reactive form for direct use by plants and animals. This shortage of fixed nitrogen has historically posed natural limits to food production and hence to human populations.

However, since the third decade of the twentieth century, the Haber-Bosch process has provided a solution. Using extremely high pressures, plus a catalyst composed mostly of iron and other critical chemicals, it became the primary procedure responsible for the production of chemical fertilizer. Today, the process is used to produce about 100 million tonnes of artificial nitrogenous fertilizer per year. Roughly 1 percent of the world's energy is used for it (Smith, 2002).

As discussed in Chapter 2, a large share of the world's crop production is fed to animals, either directly or as agro-industrial by-products. Mineral N fertilizer is applied to much of the corresponding cropland, especially in the case of high-energy crops such as maize, used in the production of concentrate feed. The gaseous emissions caused by fertilizer manufacturing should, therefore, be considered among the emissions for which the animal food chain is responsible.

About 97 percent of nitrogen fertilizers are derived from synthetically produced ammonia via the Haber-Bosch process. For economic and environmental reasons, natural gas is the fuel of choice in this manufacturing process today. Natural gas is expected to account for about one-third of global energy use in 2020, compared with only one-fifth in the mid-1990s (IFA, 2002). The ammonia industry used about 5 percent of natural gas consumption in the mid-1990s. However, ammonia production can use a wide range of energy sources. When oil and gas supplies eventually dwindle, coal can be used, and coal reserves are sufficient for well over 200 years at current production levels. In fact 60 percent of China's nitrogen fertilizer production is currently based on coal (IFA, 2002). China is an atypical case: not only is its N fertilizer production based on coal, but it is mostly produced in small and medium-sized, relatively energy-inefficient, plants. Here energy consumption per unit of N can run 20 to 25 percent higher than in plants of more recent design. One study conducted by the Chinese government estimated that energy consumption per unit of output for small plants was more than 76 percent higher than for large plants (Price *et al.*, 2000).

Before estimating the CO_2 emissions related to this energy consumption, we should try to quantify the use of fertilizer in the animal food chain. Combining fertilizer use by crop for the year 1997 (FAO, 2002) with the fraction of these crops used for feed in major N fertilizer consuming countries (FAO, 2003) shows that animal production accounts for a very substantial share of this consumption. Table 3.3 gives examples for selected countries.[4]

Table 3.3

Chemical fertilizer N used for feed and pastures in selected countries

Country	Share of total N consumption	Absolute amount
	(percentage)	*(1 000 tonnes/year)*
USA	51	4 697
China	16	2 998
France*	52	1 317
Germany*	62	1 247
Canada	55	897
UK*	70	887
Brazil	40	678
Spain	42	491
Mexico	20	263
Turkey	17	262
Argentina	29	126

* Countries with a considerable amount of N fertilized grassland.
Source: Based on FAO (2002; 2003).

Except for the Western European countries, production and consumption of chemical fertilizer is increasing in these countries. This high proportion of N fertilizer going to animal feed is largely owing to maize, which covers large areas in temperate and tropical climates and demands high doses of nitrogen fertilizer. More than half of total maize production is used as feed. Very large amounts of N fertilizer are used for maize and other animal feed, especially in nitrogen deficit areas such as North America, Southeast Asia and Western Europe. In fact maize is the crop highest in nitrogen fertilizer consumption in 18 of the 66 maize producing countries analysed (FAO, 2002). In 41 of these 66 countries maize is among the first three crops in terms of nitrogen fertilizer consumption. The projected production of maize in these countries show that its area generally expands at a rate inferior to that of production, suggesting an enhanced yield, brought about by an increase in fertilizer consumption (FAO, 2003).

Other feedcrops are also important consumers of chemical N fertilizer. Grains like barley and sorghum receive large amounts of nitrogen fertilizer. Despite the fact that some oil crops are associated with N fixing organisms themselves (see Section 3.3.1), their intensive production often makes use of nitrogen fertilizer. Such crops predominantly used as animal feed, including rapeseed, soybean and sunflower, garner considerable amounts of N-fertilizer: 20 percent of Argentina's total N fertilizer consumption is applied to production of such crops, 110 000 tonnes of N-fertilizer (for soybean alone) in Brazil and over 1.3 million tonnes in China. In addition, in a number of countries even grasslands receive a considerable amount of N fertilizer.

The countries of Table 3.3 together represent the vast majority of the world's nitrogen fertilizer use for feed production, adding a total of about 14 million tonnes of nitrogen fertilizer per year into the animal food chain. When the Commonwealth of Independent States and Oceania are added, the total rounds to around 20 percent of the annual 80 million tonnes of N fertilizer consumed worldwide. Adding in the fertilizer use that can be attributed to by-products other than oilcakes, in particular brans, may well take the total up to some 25 percent.

On the basis of these figures, the corresponding emission of carbon dioxide can be estimated. Energy requirement in modern natural gas-based systems varies between 33 and 44 gigajoules (GJ) per tonne of ammonia. Taking into consideration additional energy use in

[4] The estimates are based on the assumption of a uniform share of fertilized area in both food and feed production. This may lead to a conservative estimate, considering the large-scale, intensive production of feedcrops in these countries compared to the significant contribution of small-scale, low input production to food supply. In addition, it should be noted that these estimates do not consider the significant use of by-products other than oil cakes (brans, starch rich products, molasses, etc.). These products add to the economic value of the primary commodity, which is why some of the fertilizer applied to the original crop should be attributed to them.

Table 3.4

CO_2 emissions from the burning of fossil fuel to produce nitrogen fertilizer for feedcrops in selected countries

Country	Absolute amount of chemical N fertilizer	Energy use per tonnes fertilizer	Emission factor	Emitted CO_2
	(1 000 tonnes N fertilizer)	*(GJ/tonnes N fertilizer)*	*(tonnes C/TJ)*	*(1 000 tonnes/year)*
Argentina	126	40	17	314
Brazil	678	40	17	1 690
Mexico	263	40	17	656
Turkey	262	40	17	653
China	2 998	50	26	14 290
Spain	491	40	17	1 224
UK*	887	40	17	2 212
France*	1 317	40	17	3 284
Germany*	1 247	40	17	3 109
Canada	897	40	17	2 237
USA	4 697	40	17	11 711
Total	**14 million tonnes**			**41 million tonnes**

* Includes a considerable amount of N fertilized grassland.
Source: FAO (2002; 2003); IPCC (1997).

packaging, transport and application of fertilizer (estimated to represent an additional cost of at least 10 percent; Helsel, 1992), an upper limit of 40 GJ per tonne has been applied here. As mentioned before, energy use in the case of China is considered to be some 25 percent higher, i.e. 50 GJ per tonne of ammonia. Taking the IPCC emission factors for coal in China (26 tonnes of carbon per terajoule) and for natural gas elsewhere (17 tonnes C/TJ), estimating carbon 100 percent oxidized (officially estimated to vary between 98 and 99 percent) and applying the CO_2/C molecular weight ratio, this results in **an estimated annual emission of CO_2 of more than 40 million tonnes** (Table 3.4) at this initial stage of the animal food chain.

On-farm fossil fuel use may emit 90 million tonnes CO_2 per year

The share of energy consumption accounted for by the different stages of livestock production varies widely, depending on the intensity of livestock production (Sainz, 2003). In modern production systems the bulk of the energy is spent on production of feed, whether forage for ruminants or concentrate feed for poultry or pigs. As well as the energy used for fertilizer, important amounts of energy are also spent on seed, herbicides/pesticides, diesel for machinery (for land preparation, harvesting, transport) and electricity (irrigation pumps, drying, heating, etc.). On-farm use of fossil fuel by intensive systems produces CO_2 emissions probably even larger than those from chemical N fertilizer for feed. Sainz (2003) estimated that, during the 1980s, a typical farm in the United States spent some 35 megajoules (MJ) of energy per kilogram of carcass for chicken, 46 MJ for pigs and 51 MJ for beef, of which amounts 80 to 87 percent was spent for production.[5] A large share of this is in the form of electricity, producing much lower emissions on an energy equivalent basis than the direct use of fossil sources for energy. The share of electricity is larger for intensive monogastrics production (mainly for heating, cooling and ven-

[5] As opposed to post-harvest processing, transportation, storage and preparation. Production includes energy use for feed production and transport.

Table 3.5

On-farm energy use for agriculture in Minnesota, United States

Commodity	Minnesota ranking within USA	Crop area (10^3 km^2) head (10^6) tonnes (10^6)	Diesel (1 000 m^3 ~ 2.65–10^3 tonnes CO_2)	LPG (1 000 m^3 ~ 2.30–10^3 tonnes CO_2)	Electricity (10^6 kWh ~ 288 tonnes CO_2)	Directly emitted CO_2 (10^3 tonnes)
Corn	4	27.1	238	242	235	1 255
Soybeans	3	23.5	166	16	160	523
Wheat	3	9.1	62	6.8	67	199
Dairy (tonnes)	5	4.3 *	47	38	367	318
Swine	3	4.85	59	23	230	275
Beef	12	0.95	17	6	46	72
Turkeys (tonnes)	2	40	14	76	50	226
Sugar beets	1	1.7	46	6	45	149
Sweet corn/peas	1	0.9	9	–	5	25

Note: Reported nine commodities dominate Minnesota's agricultural output and, by extension, the state's agricultural energy use. Related CO_2 emissions based on efficiency and emission factors from the United States' Common Reporting Format report submitted to the UNFCCC in 2005.
Source: Ryan and Tiffany (1998).

tilation), which though also uses larger amounts of fossil fuel in feed transportation. However, more than half the energy expenditure during livestock production is for feed production (nearly all in the case of intensive beef operations). We have already considered the contribution of fertilizer production to the energy input for feed: in intensive systems, the combined energy-use for seed and herbicide/pesticide production and fossil fuel for machinery generally exceeds that for fertilizer production.

There are some cases where feed production does not account for the biggest share of fossil energy use. Dairy farms are an important example, as illustrated by the case of Minnesota dairy operators. Electricity is their main form of energy use. In contrast, for major staple crop farmers in the state, diesel is the dominant form of on-farm energy use, resulting in much higher CO_2 emissions (Ryan and Tiffany, 1998, presenting data for 1995). On this basis, we can suggest that the bulk of Minnesota's on-farm CO_2 emissions from energy use are also related to feed production, and exceed the emissions associated with N fertilizer use. The average maize fertilizer application (150 kg N per hectare for maize in the United States) results in emissions for Minnesota maize of about one million tonnes of CO_2, compared with 1.26 million tonnes of CO_2 from on-farm energy use for corn production (see Table 3.5). At least half the CO_2 emissions of the two dominant commodities and CO_2 sources in Minnesota (maize and soybean) can be attributed to the (intensive) livestock sector. Taken together, feed production and pig and dairy operations make the livestock sector by far the largest source of agricultural CO_2 emissions in Minnesota.

In the absence of similar estimates representative of other world regions it remains impossible to provide a reliable quantification of the global CO_2 emissions that can be attributed to on farm fossil fuel-use by the livestock sector. The energy intensity of production as well as the source of this energy vary widely. A rough indication of the fossil fuel use related emissions from intensive systems can, nevertheless, be obtained by supposing that the expected lower energy need for feed production at lower latitudes (lower energy need for corn drying for example) and the elsewhere, often lower level of mechanization, are overall compensated by a lower energy use efficiency and a lower share of relatively low CO_2 emitting sources (natural gas and electricity). Minnesota figures can then be combined with global feed production and livestock populations in intensive systems. The resulting estimate for maize only is of a magnitude similar to the emissions from manufacturing N fertilizer for use on feedcrops. As a conservative estimate, we may suggest that CO_2 emissions induced by on-farm fossil fuel use for feed production may be 50 percent higher than that from feed-dedicated N fertilizer production, i.e. some 60 million tonnes CO_2 globally. To this we must add farm emissions related directly to livestock rearing, which we may estimate at roughly 30 million tonnes of CO_2 (this figure is derived by applying Minnesota's figures to the global total of intensively-managed livestock populations, assuming that lower energy use for heating at lower latitudes is counterbalanced by lower energy efficiency and higher ventilation requirements).

On-farm fossil fuel use induced emissions in extensive systems sourcing their feed mainly from natural grasslands or crop residues can be expected to be low or even negligible in comparison to the above estimate. This is confirmed by the fact that there are large areas in developing countries, particularly in Africa and Asia, where animals are an important source of draught power, which could be considered as a CO_2 emission avoiding practice. It has been estimated that animal traction covered about half the total area cultivated in the developing countries in 1992 (Delgado *et al.*, 1999). There are no more recent estimates and it can be assumed that this share is decreasing quickly in areas with rapid mechanization, such as China or parts of India. However, draught animal power remains an important form of energy, substituting for fossil fuel combustion in many parts of the world, and in some areas, notably in West Africa, is on the increase.

Example of deforestation and shifting cultivation on steep hillside. Destruction of forests causes disastrous soil erosion in a few years – Thailand 1979

Livestock-related land use changes may emit 2.4 billion tonnes of CO_2 per year

Land use in the various parts of the world is continually changing, usually in response to competitive demand between users. Changes in land use have an impact in carbon fluxes, and many of the land-use changes involve livestock, either occupying land (as pasture or arable land for feedcrops) or releasing land for other purposes, when for example, marginal pasture land is converted to forest.

A forest contains more carbon than does a field of annual crops or pasture, and so when forests are harvested, or worse, burned, large amounts of carbon are released from the vegetation and soil to the atmosphere. The net reduction in carbon stocks is not simply equal to the net CO_2 flux from the cleared area. Reality is more complex: forest clearing can produce a complex pattern of net fluxes that change direction over time (IPCC guidelines). The calculation of carbon fluxes owing to forest conversion is, in many ways, the most complex of the emissions inventory components. Estimates of emissions from forest clearing vary because of multiple uncertainties: annual forest clearing rates, the fate of the cleared land, the amounts of carbon contained in different ecosystems, the modes by which CO_2 is released (e.g., burning or decay), and the amounts of carbon released from soils when they are disturbed.

Responses of biological systems vary over different time-scales. For example, biomass burning occurs within less than one year, while the decomposition of wood may take a decade, and loss of soil carbon may continue for several decades or even centuries. The IPCC (2001b) estimated the average annual flux owing to tropical deforestation for the decade 1980 to 1989 at 1.6±1.0 billion tonnes C as CO_2 (CO_2-C). Only about 50–60 percent of the carbon released from forest conversion in any one year was a result of the conversion and subsequent biomass burning in that year. The remainder were delayed emissions resulting from oxidation of biomass harvested in previous years (Houghton, 1991).

Clearly, estimating CO_2 emissions from land use and land-use change is far less straightforward than those related to fossil fuel combustion. It is even more difficult to attribute these emissions to a particular production sector such as livestock. However, livestock's role in deforestation is of proven importance in Latin America, the continent suffering the largest net loss of forests and resulting carbon fluxes. In Chapter 2 Latin America was identified as the region where expansion of pasture and arable land for feedcrops is strongest, mostly at the expense of forest area. The LEAD study by Wassenaar *et al.*, (2006) and Chapter 2 showed that most of the cleared area ends up as pasture and identified large areas where livestock ranching is probably a primary motive for clearing. Even if these final land uses were only one reason among many others that led to the forest clearing, animal production is certainly one of the driving forces of deforestation. The conversion of forest into pasture releases considerable amounts of carbon into the atmosphere, particularly when the area is not logged but simply burned. Cleared patches may go through several changes of land-use type. Over the 2000–2010 period, the pasture areas in Latin America are projected to expand into forest by an annual average of 2.4 million hectares – equivalent to some 65 percent of expected deforestation. If we also assume that at least half the cropland expansion into forest in Bolivia and Brazil can be attributed to providing feed for the livestock sector, this results in an additional annual deforestation for livestock of over 0.5 million hectares – giving a total for pastures plus feedcrop land, of some 3 million hectares per year.

In view of this, and of worldwide trends in extensive livestock production and in cropland for feed production (Chapter 2), we can realistically estimate that "livestock induced" emissions from deforestation amount to roughly 2.4 billion tonnes of CO_2 per year. This is based on the somewhat simplified assumption that forests are completely converted into climatically equivalent grasslands and croplands (IPCC 2001b, p. 192), combining changes in carbon density of both vegetation and soil[6] in the year of change. Though physically incorrect (it takes well over a year to reach this new status because of the "inherited", i.e. delayed emissions) the resulting emission estimate is correct provided the change process is continuous.

Other possibly important, but un-quantified, livestock-related deforestation as reported from for example Argentina (see Box 5.5 in Section 5.3.3) is excluded from this estimate.

In addition to producing CO_2 emissions, the land conversion may also negatively affect other emissions. Mosier *et al.* (2004) for example noted that upon conversion of forest to grazing land, CH_4 oxidation by soil micro-organisms is typically greatly reduced and grazing lands may even become net sources in situations where soil compaction from cattle traffic limits gas diffusion.

[6] The most recent estimates provided by this source are 194 and 122 tonnes of carbon per hectare in tropical forest, respectively for plants and soil, as opposed to 29 and 90 for tropical grassland and 3 and 122 for cropland.

Livestock-related releases from cultivated soils may total 28 million tonnes CO_2 per year

Soils are the largest carbon reservoir of the terrestrial carbon cycle. The estimated total amount of carbon stored in soils is about 1 100 to 1 600 billion tonnes (Sundquist, 1993), more than twice the carbon in living vegetation (560 billion tonnes) or in the atmosphere (750 billion tonnes). Hence even relatively small changes in carbon stored in the soil could make a significant impact on the global carbon balance (Rice, 1999).

Carbon stored in soils is the balance between the input of dead plant material and losses due to decomposition and mineralization processes. Under aerobic conditions, most of the carbon entering the soil is unstable and therefore quickly respired back to the atmosphere. Generally, less than 1 percent of the 55 billion tonnes of C entering the soil each year accumulates in more stable fractions with long mean residence times.

Human disturbance can speed up decomposition and mineralization. On the North American Great Plains, it has been estimated that approximately 50 percent of the soil organic carbon has been lost over the past 50 to 100 years of cultivation, through burning, volatilization, erosion, harvest or grazing (SCOPE 21, 1982). Similar losses have taken place in less than ten years after deforestation in tropical areas (Nye and Greenland, 1964). Most of these losses occur at the original conversion of natural cover into managed land.

Further soil carbon losses can be induced by management practices. Under appropriate management practices (such as zero tillage) agricultural soils can serve as a carbon sink and may increasingly do so in future (see Section 3.5.1). Currently, however, their role as carbon sinks is globally insignificant. As described in Chapter 2, a very large share of the production of coarse grains and oil crops in temperate regions is destined for feed use.

The vast majority of the corresponding area is under large-scale intensive management, dominated by conventional tillage practices that gradually lower the soil organic carbon content and produce significant CO_2 emissions. Given the complexity of emissions from land use and land-use changes, it is not possible to make a global estimation at an acceptable level of precision. Order-of-magnitude indications can be made by using an average loss rate from soil in a rather temperate climate with moderate to low organic matter content that is somewhere between the loss rate reported for zero and conventional tillage: Assuming an annual loss rate of 100 kg CO_2 per hectare per year (Sauvé *et al.*, 2000: covering temperate brown soil CO_2 loss, and excluding emissions originating from crop residues), the approximately 1.8 million km^2 of arable land cultivated with maize, wheat and soybean for feed would add an annual CO_2 flux of some 18 million tonnes to the livestock balance.

Tropical soils have lower average carbon content (IPCC 2001b, p. 192), and therefore lower emissions. On the other hand, the considerable expansion of large-scale feedcropping, not only into uncultivated areas, but also into previous pastureland or subsistence cropping, may increase CO_2 emission. In addition, practices such as soil liming contribute to emissions. Soil liming is a common practice in more intensively cultivated tropical areas because of soil acidity. Brazil[7] for example estimated its CO_2 emissions owing to soil liming at 8.99 million tonnes in 1994, and these have most probably increased since than. To the extent that these emissions concern cropland for feed production they should be attributed to the livestock sector. Often only crop residues and by-products are used for feeding, in which case a share of emissions corresponding to the value fraction of the commodity[8] (Chapagain and Hoekstra, 2004) should be attributed to livestock. Comparing

[7] Brazil's first national communication to the UNFCCC, 2004.

[8] The value fraction of a product is the ratio of the market value of the product to the aggregated market value of all the products obtained from the primary crop.

reported emissions from liming from national communications of various tropical countries to the UNFCCC with the importance of feed production in those countries shows that the global share of liming related emissions attributable to livestock is in the order of magnitude of Brazil's emission (0.01 billion tonnes CO_2).

Another way livestock contributes to gas emissions from cropland is through methane emissions from rice cultivation, globally recognized as an important source of methane. Much of the methane emissions from rice fields are of animal origin, because the soil bacteria are to a large extent "fed" with animal manure, an important fertilizer source (Verburg, Hugo and van der Gon, 2001). Together with the type of flooding management, the type of fertilization is the most important factor controlling methane emissions from rice cultivated areas. Organic fertilizers lead to higher emissions than mineral fertilizers. Khalil and Shearer (2005) argue that over the last two decades China achieved a substantial reduction of annual methane emissions from rice cultivation – from some 30 million tonnes per year to perhaps less than 10 million tonnes per year – mainly by replacing organic fertilizer with nitrogen-based fertilizers. However, this change can affect other gaseous emissions in the opposite way. As nitrous oxide emissions from rice fields increase, when artificial N fertilizers are used, as do carbon dioxide emissions from China's flourishing charcoal-based nitrogen fertilizer industry (see preceding section). Given that it is impossible to provide even a rough estimate of livestock's contribution to methane emissions from rice cultivation, this is not further considered in the global quantification.

Releases from livestock-induced desertification of pastures may total 100 million tonnes CO_2 per year

Livestock also play a role in desertification (see Chapters 2 and 4). Where desertification is occurring, degradation often results in reduced productivity or reduced vegetation cover, which produce a change in the carbon and nutrient stocks and cycling of the system. This seems to result in a small reduction in aboveground C stocks and a slight decline in C fixation. Despite the small, sometimes undetectable changes in aboveground biomass, total soil carbon usually declines. A recent study by Asner, Borghi and Ojeda, (2003) in Argentina also found that desertification resulted in little change in woody cover, but there was a 25 to 80 percent decline in soil organic carbon in areas with long-term grazing. Soil erosion accounts for part of this loss, but the majority stems from the non-renewal of decaying organic matter stocks, i.e. there is a significant net emission of CO_2.

Lal (2001) estimated the carbon loss as a result of desertification. Assuming a loss of 8-12 tonnes of soil carbon per hectare (Swift *et al*., 1994) on a desertified land area of 1 billion hectares (UNEP, 1991), the total historic loss would amount to 8–12 billion tonnes of soil carbon. Similarly, degradation of aboveground vegetation has led to an estimated carbon loss of 10–16 tonnes per hectare – a historic total of 10–16 billion tonnes. Thus, the total C loss as a consequence of desertification may be 18–28 billion tonnes of carbon (FAO, 2004b). Livestock's contribution to this total is difficult to estimate, but it is undoubtedly high: livestock occupies about two-thirds of the global dry land area, and the rate of desertification has been estimated to be higher under pasture than under other land uses (3.2 million hectares per year against 2.5 million hectares per year for cropland, UNEP, 1991). Considering only soil carbon loss (i.e. about 10 tonnes of carbon per hectare), pasture desertification-induced oxidation of carbon would result in CO_2 emissions in the order of 100 million tonnes of CO_2 per year.

Another, largely unknown, influence on the fate of soil carbon is the feedback effect of climate change. In higher latitude cropland zones, global warming is expected to increase yields by virtue of longer growing seasons and CO_2 fertilization (Cantagallo, Chimenti and Hall, 1997; Travasso *et al*., 1999). At the same time, however, global

Box 3.2 The many climatic faces of the burning of tropical savannah

Burning is common in establishing and managing of pastures, tropical rain forests and savannah regions and grasslands worldwide (Crutzen and Andreae, 1990; Reich *et al.*, 2001). Fire removes ungrazed grass, straw and litter, stimulates fresh growth, and can control the density of woody plants (trees and shrubs). As many grass species are more fire-tolerant than tree species (especially seedlings and saplings), burning can determine the balance between grass cover and ligneous vegetation. Fires stimulate the growth of perennial grasses in savannahs and provide nutritious re-growth for livestock. Controlled burning prevents uncontrolled, and possibly, more destructive fires and consumes the combustible lower layer at an appropriate humidity stage. Burning involves little or no cost. It is also used at a small scale to maintain biodiversity (wildlife habitats) in protected areas.

The environmental consequences of rangeland and grassland fires depend on the environmental context and conditions of application. Controlled burning in tropical savannah areas has significant environmental impact, because of the large area concerned and the relatively low level of control. Large areas of savannah in the humid and subhumid tropics are burned every year for rangeland management. In 2000, burning affected some 4 million km^2. More than two-thirds of this occurred in the tropics and sub-tropics (Tansey *et al.*, 2004). Globally about three quarters of this burning took place outside forests. Savannah burning represented some 85 percent of the area burned in Latin American fires 2000, 60 percent in Africa, nearly 80 percent in Australia.

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Hunter set fire to forest areas to drive out a species of rodent that will be killed for food. Herdsmen and hunters together benefit from the results.

Usually, savannah burning is not considered to result in net CO_2 emissions, since emitted amounts of carbon dioxide released in burning are re-captured in grass re-growth. As well as CO_2, biomass burning releases important amounts of other globally relevant trace gases (NO_x, CO, and CH_4) and aerosols (Crutzen and Andreae, 1990; Scholes and Andreae, 2000). Climate effects include the formation of photochemical smog, hydrocarbons, and NO_x. Many of the emitted elements lead to the production of tropospheric ozone (Vet, 1995; Crutzen and Goldammer, 1993), which is another important greenhouse gas influencing the atmosphere's oxidizing capacity, while bromine, released in significant amounts from savannah fires, decreases stratospheric ozone (Vet, 1995; ADB, 2001).

Smoke plumes may be redistributed locally, transported throughout the lower troposphere, or entrained in large-scale circulation patterns in the mid and upper troposphere. Often fires in convection areas take the elements high into the atmosphere, creating increased potential for climate change. Satellite observations have found large areas with high O_3 and CO levels over Africa, South America and the tropical Atlantic and Indian Oceans (Thompson *et al.*, 2001).

Aerosols produced by the burning of pasture biomass dominate the atmospheric concentration of aerosols over the Amazon basin and Africa (Scholes and Andreae, 2000; Artaxo *et al.*, 2002). Concentrations of aerosol particles are highly seasonal. An obvious peak in the dry (burning) season, which contributes to cooling both through increasing atmospheric scattering of incoming light and the supply of cloud condensation nuclei. High concentrations of cloud condensation nuclei from the burning of biomass stimulate rainfall production and affect large-scale climate dynamics (Andreae and Crutzen, 1997).

warming may also accelerate decomposition of carbon already stored in soils (Jenkinson,1991; MacDonald, Randlett and Zalc, 1999; Niklinska, Maryanski and Laskowski, 1999; Scholes *et al.*, 1999). Although much work remains to be done in quantifying the CO_2 fertilization effect in cropland, van Ginkel, Whitmore and Gorissen, (1999) estimate the magnitude of this effect (at current rates of increase of CO_2 in the atmosphere) at a net absorption of 0.036 tonnes of carbon per hectare per year in temperate grassland, even after the effect of rising temperature on decomposition is deducted. Recent research indicates that the magnitude of the temperature rise on the acceleration of decay may be stronger, with already very significant net losses over the last decades in temperate regions (Bellamy *et al.*, 2005; Schulze and Freibauer, 2005). Both scenarios may prove true, resulting in a shift of carbon from soils to vegetation – i.e. a shift towards more fragile ecosystems, as found currently in more tropical regions.

3.2.2 Carbon emissions from livestock rearing

Respiration by livestock is not a net source of CO_2

Humans and livestock now account for about a quarter of the total terrestrial animal biomass.[9] Based on animal numbers and liveweights, the total livestock biomass amounts to some 0.7 billion tonnes (Table 3.6; FAO, 2005b).

How much do these animals contribute to greenhouse gas emissions? According to the function established by Muller and Schneider (1985, cited by Ni *et al.*, 1999), applied to standing stocks per country and species (with country specific liveweight), the carbon dioxide from the respiratory process of livestock amount to some 3 billion tonnes of CO_2 (see Table 3.6) or 0.8 billion tonnes of carbon. In general, because of lower offtake rates and therefore higher inventories, ruminants have higher emissions relative to their output. Cattle alone account for more than half of the total carbon dioxide emissions from respiration.

However, emissions from livestock respiration are part of a rapidly cycling biological system, where the plant matter consumed was itself created through the conversion of atmospheric CO_2 into organic compounds. Since the emitted and absorbed quantities are considered to be equivalent, livestock respiration is not considered to be a net source under the Kyoto Protocol. Indeed, since part of the carbon consumed is stored in the live tissue of the growing animal, a growing global herd could even be considered a carbon sink. The standing stock livestock biomass increased significantly over the last decades (from about 428 million tonnes in 1961 to around 699 million tonnes in 2002). This continuing growth (see Chapter 1) could be considered as a carbon sequestration process (roughly estimated at 1 or 2 million tonnes carbon per year). However, this is more than offset by methane emissions which have increased correspondingly.

The equilibrium of the biological cycle is, however, disrupted in the case of overgrazing or bad management of feedcrops. The resulting land degradation is a sign of *decreasing* re-absorption of atmospheric CO_2 by vegetation re-growth. In certain regions the related net CO_2 loss may be significant.

Methane released from enteric fermentation may total 86 million tonnes per year

Globally, livestock are the most important source of anthropogenic methane emissions. Among domesticated livestock, ruminant animals (cattle, buffaloes, sheep, goats and camels) produce significant amounts of methane as part of their normal digestive processes. In the rumen, or large fore-stomach, of these animals, microbial fermentation converts fibrous feed into products that can be digested and utilized by the animal. This microbial fermentation process, referred to

[9] Based on SCOPE 13 (Bolin *et al.*, 1979), with human population updated to today's total of some 6.5 billion.

Table 3.6

Livestock numbers (2002) and estimated carbon dioxide emissions from respiration

Species	World total	Biomass	Carbon dioxide emissions
	(million head)	*(million tonnes liveweight)*	*(million tonnes CO_2)*
Cattle and buffaloes	1 496	501	1 906
Small ruminants	1 784	47.3	514
Camels	19	5.3	18
Horses	55	18.6	71
Pigs	933	92.8	590
Poultry[1]	17 437	33.0	61
Total[2]		**699**	**3 161**

[1] Chicken, ducks, turkey and geese.
[2] Includes also rabbits.
Source: FAO (2006b); own calculations.

as enteric fermentation, produces methane as a by-product, which is exhaled by the animal. Methane is also produced in smaller quantities by the digestive processes of other animals, including humans (US-EPA, 2005).

There are significant spatial variations in methane emissions from enteric fermentation. In Brazil, methane emission from enteric fermentation totalled 9.4 million tonnes in 1994 - 93 percent of agricultural emissions and 72 percent of the country's total emissions of methane. Over 80 percent of this originated from beef cattle (Ministério da Ciência e Tecnologia - EMBRAPA report, 2002). In the United States methane from enteric fermentation totalled 5.5 million tonnes in 2002, again overwhelmingly originating from beef and dairy cattle. This was 71 percent of all agricultural emissions and 19 percent of the country's total emissions (US-EPA, 2004).

This variation reflects the fact that levels of methane emission are determined by the production system and regional characteristics. They are affected by energy intake and several other animal and diet factors (quantity and quality of feed, animal body weight, age and amount of exercise). It varies among animal species and among individuals of the same species. Therefore, assessing methane emission from enteric fermentation in any particular country requires a detailed description of the livestock population (species, age and productivity categories), combined with information on the daily feed intake and the feed's methane conversion rate (IPCC revised guidelines). As many countries do not possess such detailed information, an approach based on standard emission factors is generally used in emission reporting.

Methane emissions from enteric fermentation will change as production systems change and move towards higher feed use and increased productivity. We have attempted a global estimate of total methane emissions from enteric fermentation in the livestock sector. Annex 3.2 details the findings of our assessment, compar-

Dairy cattle feeding on fodder in open stable. La Loma, Lerdo, Durango – Mexico 1990

Table 3.7

Global methane emissions from enteric fermentation in 2004

Region/country	Emissions (million tonnes CH_4 per year by source)					
	Dairy cattle	Other cattle	Buffaloes	Sheep and goats	Pigs	Total
Sub-Saharan Africa	2.30	7.47	0.00	1.82	0.02	**11.61**
Asia *	0.84	3.83	2.40	0.88	0.07	**8.02**
India	1.70	3.94	5.25	0.91	0.01	**11.82**
China	0.49	5.12	1.25	1.51	0.48	**8.85**
Central and South America	3.36	17.09	0.06	0.58	0.08	**21.17**
West Asia and North Africa	0.98	1.16	0.24	1.20	0.00	**3.58**
North America	1.02	3.85	0.00	0.06	0.11	**5.05**
Western Europe	2.19	2.31	0.01	0.98	0.20	**5.70**
Oceania and Japan	0.71	1.80	0.00	0.73	0.02	**3.26**
Eastern Europe and CIS	1.99	2.96	0.02	0.59	0.10	**5.66**
Other developed	0.11	0.62	0.00	0.18	0.00	**0.91**
Total	**15.69**	**50.16**	**9.23**	**9.44**	**1.11**	**85.63**
Livestock Production System						
Grazing	4.73	21.89	0.00	2.95	0.00	**29.58**
Mixed	10.96	27.53	9.23	6.50	0.80	**55.02**
Industrial	0.00	0.73	0.00	0.00	0.30	**1.04**

* Excludes China and India.
Source: see Annex 3.2, own calculations.

ing IPCC Tier 1 default emission factors with region-specific emission factors. Applying these emission factors to the livestock numbers in each production system gives an estimate for total global emissions of methane from enteric fermentation 86 million tonnes CH_4 annually. This is not far from the global estimate from the United States Environmental Protection Agency (US-EPA, 2005), of about 80 million tonnes of methane annually. The regional distribution of such methane emission is illustrated by Map 33 (Annex 1). This is an updated and more precise estimate than previous such attempts (Bowman *et al.*, 2000; Methane emission map published by UNEP-GRID, Lerner, Matthews and Fung, 1988) and also provides production-system specific estimates. Table 3.7 summarizes these results. The relative global importance of mixed systems compared to grazing systems reflects the fact that about two-thirds of all ruminants are held in mixed systems.

Methane released from animal manure may total 18 million tonnes per year

The anaerobic decomposition of organic material in livestock manure also releases methane. This occurs mostly when manure is managed in liquid form, such as in lagoons or holding tanks. Lagoon systems are typical for most large-scale pig operations over most of the world (except in Europe). These systems are also used in large dairy operations in North America and in some developing countries, for example Brazil. Manure deposited on fields and pastures, or otherwise handled in a dry form, does not produce significant amounts of methane.

Methane emissions from livestock manure are influenced by a number of factors that affect the growth of the bacteria responsible for methane formation, including ambient temperature, moisture and storage time. The amount of methane produced also depends on the energy content of manure, which is determined to a

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State of the art lagoon waste management system for a 900 head hog farm. The facility is completely automated and temperature controlled – United States 2002

large extent by livestock diet. Not only do greater amounts of manure lead to more CH_4 being emitted, but higher energy feed also produces manure with more volatile solids, increasing the substrate from which CH_4 is produced. However, this impact is somewhat offset by the possibility of achieving higher digestibility in feeds, and thus less wasted energy (USDA, 2004).

Globally, methane emissions from anaerobic decomposition of manure have been estimated to total just over 10 million tonnes, or some 4 percent of global anthropogenic methane emissions (US-EPA, 2005). Although of much lesser magnitude than emissions from enteric fermentation, emissions from manure are much higher than those originating from burning residues and similar to the lower estimate of the badly known emissions originating from rice cultivation. The United States has the highest emission from manure (close to 1.9 million tonnes, United States inventory 2004), followed by the EU. As a species, pig production contributes the largest share, followed by dairy. Developing countries such as China and India would not be very far behind, the latter in particular exhibiting a strong increase. The default emission factors currently used in country reporting to the UNFCCC do not reflect such strong changes in the global livestock sector. For example, Brazil's country report to the UNFCCC (Ministry of Science and Technology, 2004) mentions a significant emission from manure of 0.38 million tonnes in 1994, which would originate mainly from dairy and beef cattle. However, Brazil also has a very strong industrial pig production sector, where some 95 percent of manure is held in open tanks for several months before application (EMBRAPA, personal communication).

Hence, a new assessment of emission factors similar to the one presented in the preceding section was essential and is presented in Annex 3.3. Applying these new emission factors to the animal population figures specific to each production system, we arrive at a total annual global emission of methane from manure decomposition of 17.5 million tonnes of CH_4. This is substantially higher than existing estimates.

Table 3.8 summarizes the results by species,

Table 3.8

Global methane emissions from manure management in 2004

Region/country	Emissions (million tonnes CH_4 per year by source)						
	Dairy cattle	Other cattle	Buffalo	Sheep and goats	Pigs	Poultry	Total
Sub-Saharan Africa	0.10	0.32	0.00	0.08	0.03	0.04	**0.57**
Asia *	0.31	0.08	0.09	0.03	0.50	0.13	**1.14**
India	0.20	0.34	0.19	0.04	0.17	0.01	**0.95**
China	0.08	0.11	0.05	0.05	3.43	0.14	**3.84**
Central and South America	0.10	0.36	0.00	0.02	0.74	0.19	**1.41**
West Asia and North Africa	0.06	0.09	0.01	0.05	0.00	0.11	**0.32**
North America	0.52	1.05	0.00	0.00	1.65	0.16	**3.39**
Western Europe	1.16	1.29	0.00	0.02	1.52	0.09	**4.08**
Oceania and Japan	0.08	0.11	0.00	0.03	0.10	0.03	**0.35**
Eastern Europe and CIS	0.46	0.65	0.00	0.01	0.19	0.06	**1.38**
Other developed	0.01	0.03	0.00	0.01	0.04	0.02	**0.11**
Global Total	**3.08**	**4.41**	**0.34**	**0.34**	**8.38**	**0.97**	**17.52**
Livestock Production System							
Grazing	0.15	0.50	0.00	0.12	0.00	0.00	**0.77**
Mixed	2.93	3.89	0.34	0.23	4.58	0.31	**12.27**
Industrial	0.00	0.02	0.00	0.00	3.80	0.67	**4.48**

* Excludes China and India.
Source: see Annex 3.3, own calculations.

by region and by farming system. The distribution by species and production system is also illustrated in Maps 16, 17, 18 and 19 (Annex 1). China has the largest country-level methane emission from manure in the world, mainly from pigs. At a global level, emissions from pig manure represent almost half of total livestock manure emissions. Just over a quarter of the total methane emission from managed manure originates from industrial systems.

3.2.3 Carbon emissions from livestock processing and refrigerated transport

A number of studies have been conducted to quantify the energy costs of processing animals for meat and other products, and to identify potential areas for energy savings (Sainz, 2003). The variability among enterprises is very wide, so it is difficult to generalize. For example, Ward, Knox and Hobson, (1977) reported energy costs of beef processing in Colorado ranging from 0.84 to 5.02 million joules per kilogram of live weight. Sainz (2003) produced indicative values for the energy costs of processing, given in Table 3.9.

CO_2 emissions from livestock processing may total several tens of million tonnes per year

To obtain a global estimate of emissions from processing, these indicative energy use factors could be combined with estimates of the world's livestock production from market-oriented intensive systems (Chapter 2). However, besides their questionable global validity, it is highly uncertain what the source of this energy is and how this varies throughout the world. Since mostly products from intensive systems are being processed, the above case of Minnesota (Section 3.2.1 on *on-farm fossil fuel use* and Table 3.5) constitutes an interesting example of energy use for processing, as well as a breakdown into energy sources (Table 3.13). Diesel use here is mainly for transport of products

Table 3.9

Indicative energy costs for processing

Product	Fossil energy cost	Units	Source
Poultry meat	2.59	MJ-kg^{-1} live wt	Whitehead and Shupe, 1979
Eggs	6.12	MJ-dozen^{-1}	OECD, 1982
Pork-fresh	3.76	MJ-kg^{-1} carcass	Singh, 1986
Pork-processed meats	6.30	MJ-kg^{-1} meat	Singh, 1986
Sheep meat	10.4	MJ-kg^{-1} carcass	McChesney *et al.*, 1982
Sheep meat-frozen	0.432	MJ-kg^{-1} meat	Unklesbay and Unklesbay, 1982
Beef	4.37	MJ-kg^{-1} carcass	Poulsen, 1986
Beef-frozen	0.432	MJ-kg^{-1} meat	Unklesbay and Unklesbay, 1982
Milk	1.12	MJ-kg^{-1}	Miller, 1986
Cheese, butter, whey powder	1.49	MJ-kg^{-1}	Miller, 1986
Milk powder, butter	2.62	MJ-kg^{-1}	Miller, 1986

Source: Sainz (2003).

to the processing facilities. Transport-related emissions for milk are high, owing to large volumes and low utilization of transport capacity. In addition, large amounts of energy are used to pasteurize milk and transform it into cheese and dried milk, making the dairy sector responsible for the second highest CO_2 emissions from food processing in Minnesota. The largest emissions result from soybean processing and are a result of physical and chemical methods to separate the crude soy oil and soybean meal from the raw beans. Considering the value fractions of these two commodities (see Chapagain and Hoekstra, 2004) some two-thirds of these soy-processing emissions can be attributed to the livestock sector. Thus, the majority of CO_2 emissions related to energy consumption from processing Minnesota's agricultural production can be ascribed to the livestock sector.

Minnesota can be considered a "hotspot" because of its CO_2 emissions from livestock processing and cannot, in light of the above remarks on the variability of energy efficiency and sources, be used as a basis for deriving a global estimate. Still, considering also Table 3.10, it indicates that the total animal product and feed processing related emission of the United States would be in the order of a few million tonnes CO_2. Therefore, the probable order of magnitude for the emission level related to global animal-product processing would be several tens of million tonnes CO_2.

CO_2 emissions from transport of livestock products may exceed 0.8 million tonnes per year

The last element of the food chain to be considered in this review of the carbon cycle is the one that links the elements of the production chain and delivers the product to retailers and consumers, i.e. transport. In many instances transport is over short distances, as in the case of milk collection cited above. Increasingly the steps in the chain are separated over long distances (see Chapter 2), which makes transport a significant source of greenhouse gas emissions.

Transport occurs mainly at two key stages: delivery of (processed) feed to animal production sites and delivery of animal products to consumer markets. Large amounts of bulky raw ingredients for concentrate feed are shipped around the world (Chapter 2). These long-distance flows add significant CO_2 emissions to the livestock balance. One of the most notable long-distance feed trade flows is for soybean, which is also the largest traded volume among feed

Table 3.10

Energy use for processing agricultural products in Minnesota, in United States in 1995

Commodity	Production[1]	Diesel	Natural gas	Electricity	Emitted CO_2
	(10^6 tonnes)	*(1000 m^3)*	*(10^6 m^3)*	*(10^6 kWh)*	*(10^3 tonnes)*
Corn	22.2	41	54	48	226
Soybeans	6.4	23	278	196	648
Wheat	2.7	19	–	125	86
Dairy	4.3	36	207	162	537
Swine	0.9	7	21	75	80
Beef	0.7	2.5	15	55	51
Turkeys	0.4	1.8	10	36	34
Sugar beets[2]	7.4	19	125	68	309
Sweet corn/peas	1.0	6	8	29	40

[1] Commodities: unshelled corn ears, milk, live animal weight. 51 percent of milk is made into cheese, 35 percent is dried, and 14 percent is used as liquid for bottling.
[2] Beet processing required an additional 440 thousand tonnes of coal.
1 000 m^3 diesel ~ $2.65 \cdot 10^3$ tonnes CO_2; 10^6 m^3 natural gas ~ $1.91 \cdot 10^3$ tonnes CO_2; 10^6 kWh ~ 288 tonnes CO_2
Source: Ryan and Tiffany (1998). See also table 3.5. Related CO_2 emissions based on efficiency and emission factors from the United States' Common Reporting Format report submitted to the UNFCCC in 2005.

ingredients, as well as the one with the strongest increase. Among soybean (cake) trade flows the one from Brazil to Europe is of a particularly important volume. Cederberg and Flysjö (2004) studied the energy cost of shipping soybean cake from the Mato Grosso to Swedish dairy farms: shipping one tonne requires some 2 900 MJ, of which 70 percent results from ocean transport. Applying this energy need to the annual soybean cake shipped from Brazil to Europe, combined with the IPCC emission factor for ocean vessel engines, results in an annual emission of some 32 thousand tonnes of CO_2.

While there are a large number of trade flows, we can take pig, poultry and bovine meat to represent the emissions induced by fossil energy use for shipping animal products around the world. The figures presented in Table 15, Annex 2 are the result of combining traded volumes (FAO, accessed December 2005) with respective distances, vessel capacities and speeds, fuel use of main engine and auxiliary power generators for refrigeration, and their respective emission factors (IPCC, 1997).

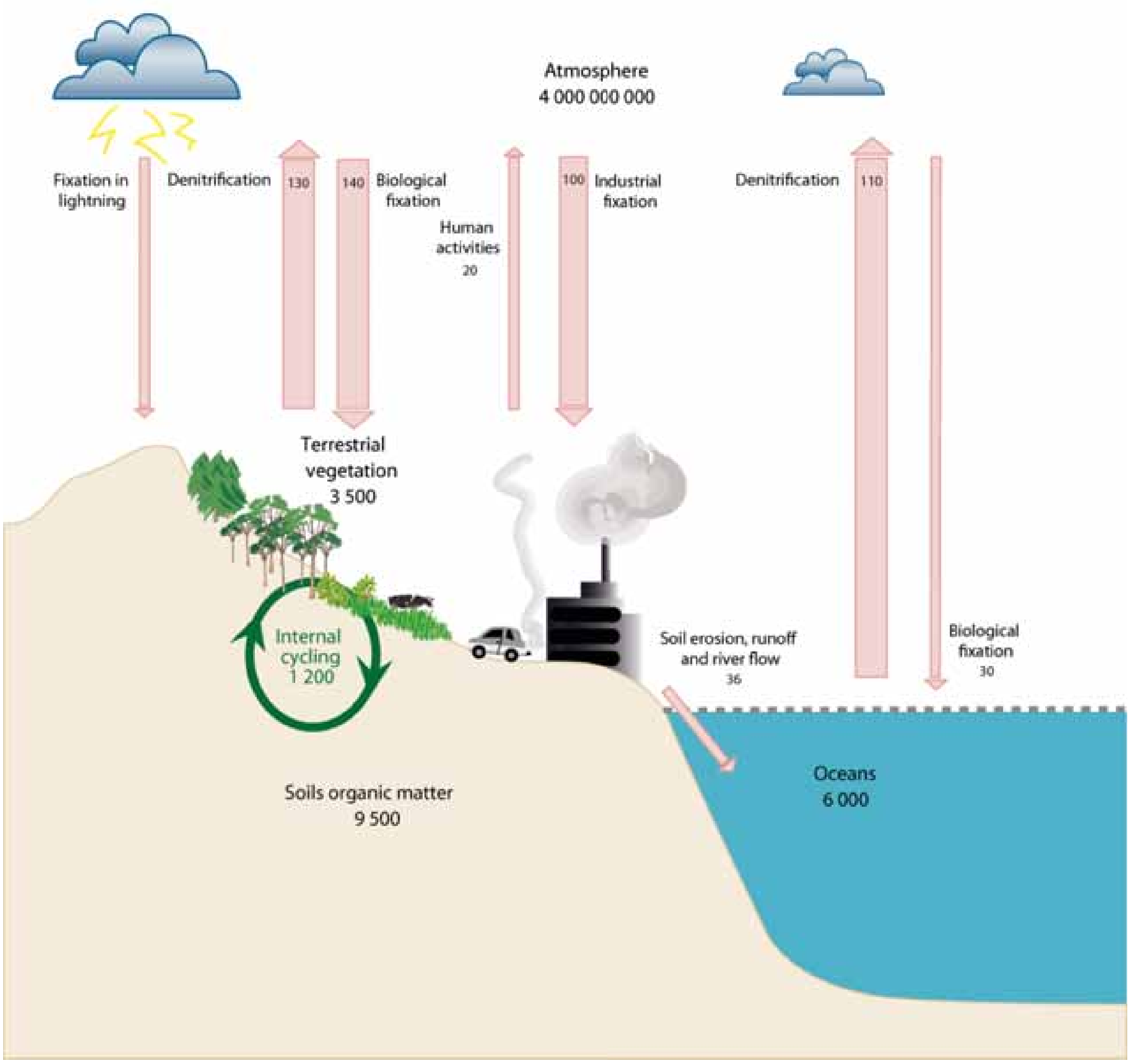
These flows represent some 60 percent of international meat trade. Annually they produce some 500 thousand tonnes of CO_2. This represents more than 60 percent of total CO_2 emissions induced by meat-related sea transport, because the trade flow selection is biased towards the long distance exchange. On the other hand, surface transport to and from the harbour has not been considered. Assuming, for simplicity, that the latter two effects compensate each other, the total annual meat transport-induced CO_2 emission would be in the order of 800-850 thousand tonnes of CO_2.

3.3 Livestock in the nitrogen cycle

Nitrogen is an essential element for life and plays a central role in the organization and functioning of the world's ecosystems. In many terrestrial and aquatic ecosystems, the availability of nitrogen is a key factor determining the nature and diversity of plant life, the population dynamics of both grazing animals and their predators, and vital ecological processes such as plant productivity and the cycling of carbon and soil minerals (Vitousek *et al.*, 1997).

The natural carbon cycle is characterized by

Figure 3.2 **The nitrogen cycle**



Source: Porter and Botkin (1999).

large fossil terrestrial and aquatic pools, and an atmospheric form that is easily assimilated by plants. The nitrogen cycle is quite different: diatomic nitrogen (N_2) in the atmosphere is the sole stable (and very large) pool, making up some 78 percent of the atmosphere (see Figure 3.2).

Although nitrogen is required by all organisms to survive and grow, this pool is largely unavailable to them under natural conditions. For most organisms this nutrient is supplied via the tissues of living and dead organisms, which is why many ecosystems of the world are limited by nitrogen.

The few organisms able to assimilate atmospheric N_2 are the basis of the natural N cycle of modest intensity (relative to that of the C cycle), resulting in the creation of dynamic pools in organic matter and aquatic resources. Generally put, nitrogen is removed from the atmosphere by soil micro-organisms, such as the nitrogen-fixing bacteria that colonize the roots of leguminous plants. These bacteria convert it into forms (so-called reactive nitrogen, Nr, in essence all N compounds other than N_2) such as ammonia (NH_3), which can then be used by the plants. This process is called nitrogen fixation. Meanwhile, other micro-organisms remove nitrogen from the soil and put it back into the atmosphere. This process, called denitrification, returns N to the atmosphere in various forms, primarily N_2. In addition, denitrification produces the greenhouse gas nitrous oxide.

The human impact on the nitrogen cycle

The modest capability of natural ecosystems to drive the N cycle constituted a major hurdle in satisfying the food needs of growing populations (Galloway *et al.*, 2004). The historical increases of legume, rice and soybean cultivation increased N fixation, but the needs of large populations could only be met after the invention of the Haber-Bosch process in the first decade of the twentieth century, to transform N_2 into mineral fertilizers (see section on feed sourcing).

In view of the modest natural cycling intensity, additions of chemical N fertilizers had dramatic effects. It has been estimated that humans have already doubled the natural rate of nitrogen entering the land-based nitrogen cycle and this rate is continuing to grow (Vitousek *et al.*, 1997). Synthetic fertilizers now provide about 40 percent of all the nitrogen taken up by crops (Smil, 2001). Unfortunately crop, and especially animal, production uses this additional resource at a rather low efficiency of about 50 percent. The rest is estimated to enter the so-called *nitrogen cascade* (Galloway *et al.*, 2003) and is transported downstream or downwind where the nitrogen can have a sequence of effects on ecosystems and people. Excessive nitrogen additions can pollute ecosystems and alter both their ecological functioning and the living communities they support.

What poses a problem to the atmosphere is that human intervention in the nitrogen cycle has changed the balance of N species in the atmosphere and other reservoirs. Non-reactive molecular nitrogen is neither a greenhouse gas nor an air polluter. However, human activities return much of it in the form of reactive nitrogen species which either is a greenhouse gas or an air polluter. Nitrous oxide is very persistent in the atmosphere where it may last for up to 150 years. In addition to its role in global warming, N_2O is also involved in the depletion of the ozone layer, which protects the biosphere from the harmful effects of solar ultraviolet radiation (Bolin *et al.*, 1981). Doubling the concentration of N_2O in the atmosphere would result in an estimated 10 percent decrease in the ozone layer, which in turn would increase the ultraviolet radiation reaching the earth by 20 percent.

The atmospheric concentration of nitrous oxide has steadily increased since the beginning of the industrial era and is now 16 percent (46 ppb) larger than in 1750 (IPCC, 2001b). Natural sources of N_2O are estimated to emit approximately 10 million tonnes N/yr, with soils contributing about 65 percent and oceans about 30 percent. According to recent estimates, N_2O emissions from anthropogenic sources (agriculture, biomass burning, industrial activities and livestock management) amount to approximately 7–8 million tonnes N/yr (van Aardenne *et al.*, 2001; Mosier *et al.*, 2004). According to these estimates, 70 percent of this results from agriculture, both crop and livestock production. Anthropogenic NO emissions also increased substantially. Although it is not a greenhouse gas (and, therefore, is not further considered in this section), NO is involved in the formation process of ozone, which is a greenhouse gas.

Though quickly re-deposited (hours to days), annual atmospheric emissions of air-polluting ammonia (NH_3) increased from some 18.8 million tonnes N at the end of the 19th century to about 56.7 million tonnes in the early 1990s. They are projected to rise to 116 million tonnes N/yr by 2050, giving rise to considerable air pollution in a number of world regions (Galloway *et al*., 2004). This would be almost entirely caused by food production and particularly by animal manure.

Besides increased fertilizer use and agricultural nitrogen fixation, the enhanced N_2O emissions from agricultural and natural ecosystems are also caused by increasing N deposition (mainly of ammonia). Whereas terrestrial ecosystems in the northern hemisphere are limited by nitrogen, tropical ecosystems, currently an important source of N_2O (and NO), are often limited by phosphorus. Nitrogen fertilizer inputs into these phosphorus-limited ecosystems generate NO and N_2O fluxes that are 10 to 100 times greater than the same fertilizer addition to N-limited ecosystems (Hall and Matson, 1999).

Soil N_2O emissions are also regulated by temperature and soil moisture and so are likely to respond to climate changes (Frolking *et al*., 1998). In fact, chemical processes involving nitrous oxides are extremely complex (Mosier *et al*., 2004). Nitrification – the oxidation of ammonia to nitrite and then nitrate – occurs in essentially all terrestrial, aquatic and sedimentary ecosystems and is accomplished by specialized bacteria. Denitrification, the microbial reduction of nitrate or nitrite to gaseous nitrogen with NO and N_2O as intermediate reduction compounds, is performed by a diverse and also widely distributed group of aerobic, heterotrophic bacteria.

The main use of ammonia today is in fertilizers, produced from non-reactive molecular nitrogen, part of which directly volatilizes. The largest atmospheric ammonia emission overall comes from the decay of organic matter in soils. The quantity of ammonia that actually escapes from soils into the atmosphere is uncertain; but is estimated at around 50 million tonnes per year (Chameides and Perdue, 1997). As much as 23 million tonnes N of ammonia are produced each year by domesticated animals, while wild animals contribute roughly 3 million tonnes N/yr and human waste adds 2 million tonnes N/yr. Ammonia dissolves easily in water, and is very reactive with acid compounds. Therefore, once in the atmosphere, ammonia is absorbed by water and reacts with acids to form salts. These salts are deposited again on the soil within hours to days (Galloway *et al*., 2003) and they in turn can have an impact on ecosystems.

3.3.1 Nitrogen emissions from feed-related fertilizer

The estimated global NH_3 volatilization loss from synthetic N fertilizer use in the mid-1990s totalled about 11 million tonnes N per year. Of this 0.27 million tonnes emanated from fertilized grasslands, 8.7 million tonnes from rainfed crops and 2.3 million tonnes from wetland rice (FAO/IFA, 2001), estimating emissions in 1995). Most of this occurs in the developing countries (8.6 million tonnes N), nearly half of which in China. Average N losses as ammonia from synthetic fertilizer use is more than twice as high (18 percent) in developing countries than in developed and transition countries (7 percent). Most of this difference in loss rates is resulting from higher temperatures and the dominant use of urea and ammonium bicarbonate in the developing world.

In developing countries about 50 percent of the nitrogen fertilizer used is in the form of urea (FAO/IFA, 2001). Bouwman *et al*. (1997) estimate that NH_3 emission losses from urea may be 25 percent in tropical regions and 15 percent in temperate climates. In addition, NH_3 emissions may be higher in wetland rice cultivation than in dryland fields. In China, 40–50 percent of the nitrogen fertilizer used is in the form of ammonium bicarbonate, which is highly volatile. The NH_3 loss from ammonium bicarbonate may be 30 percent on average in the tropics and 20 percent in temperate zones. By contrast, the NH_3 loss from injected anhydrous ammonia, widely used in the United States, is only 4 percent (Bouwman *et al*., 1997).

What share of direct emissions from fertilizer can we attribute to livestock? As we have seen, a large share of the world's crop production is fed to animals and mineral fertilizer is applied to much of the corresponding cropland. Intensively managed grasslands also receive a significant portion of mineral fertilizer. In Section 3.2.1 we estimated that 20 to 25 percent of mineral fertilizer use (about 20 million tonnes N) can be ascribed to feed production for the livestock sector. Assuming that the low loss rates of an important "fertilizer for feed" user such as the United States is compensated by high loss rates in South and East Asia, the average mineral fertilizer NH_3 volatilization loss rate of 14 percent (FAO/IFA, 2001) can be applied. On this basis, livestock production can be considered responsible for a global NH_3 volatilization from mineral fertilizer of 3.1 million tonnes NH_3-N (tonnes of nitrogen in ammonia form) per year.

Turning now to N_2O, the level of emissions from mineral N fertilizer application depends on the mode and timing of fertilizer application. N_2O emissions for major world regions can be estimated using the FAO/IFA (2001) model. Nitrous oxide emissions amount to 1.25 ± 1 percent of the nitrogen applied. This estimate is the average for all fertilizer types, as proposed by Bouwman (1995) and adopted by IPCC (1997). Emission rates also vary from one fertilizer type to another. The FAO/IFA (2001) calculations result in a mineral fertilizer N_2O-N loss rate of 1 percent. Under the same assumptions as for NH_3 above, livestock production can be considered responsible for a global N_2O emission from mineral fertilizer of 0.2 million tonne N_2O-N per year.

There is also N_2O emission from leguminous feedcrops, even though they do not generally receive N fertilizer because the rhizobia in their root nodules fix nitrogen that can be used by the plant. Studies have demonstrated that such crops show N_2O emissions of the same level as those of fertilized non-leguminous crops. Considering the world area of soybean and pulses, and the share of production used for feed, gives a total of some 75 million hectares in 2002 (FAO, 2006b). This would amount to another 0.2 million tonnes of N_2O-N per year. Adding alfalfa and clovers would probably about double this figure, although there are no global estimates of their cultivated areas. Russelle and Birr (2004) for example show that soybean and alfalfa together harvest some 2.9 million tonne of fixed N in the Mississippi River Basin, with the N_2 fixation rate of alfalfa being nearly twice as high as that of soybean (see also a review in Smil, 1999). It seems therefore probable that livestock production can be considered responsible for a total N_2O-N emission from soils under leguminous crops exceeding 0.5 million tonnes per year and a total emission from feedcropping exceeding 0.7 million tonne N_2O-N.

3.3.2 Emissions from aquatic sources following chemical fertilizer use

The above direct cropland emissions represent some 10 to 15 percent of the anthropogenic, added reactive N (mineral fertilizer and cultivation-induced biological nitrogen fixation – BNF). Unfortunately, a very large share of the remaining N is not incorporated in the harvested plant tissue nor stored in the soil. Net changes in the organically bound nitrogen pool of the world's agricultural soils are very small and may be positive or negative (plus or minus 4 million tonnes N, see Smil, 1999). Soils in some regions have significant gains whereas poorly managed soils in other regions suffer large losses.

As Von Liebig noted back in 1840 (cited in Smil, 2002) one of agriculture's main objectives is to produce digestible N, so cropping aims to accumulate as much N as possible in the harvested product. But even modern agriculture involves substantial losses – N efficiency in global crop production is estimated to be only 50 or 60 percent (Smil, 1999; van der Hoek, 1998). Reworking these estimates to express efficiency as the amount of N harvested from the world's cropland with respect to the annual N input,[10] results in an even lower efficiency of some 40 percent.

This result is affected by animal manure, which has a relatively high loss rate as compared to mineral fertilizer (see following section). Mineral fertilizer is more completely absorbed, depending on the fertilizer application rate and the type of mineral fertilizer. The most efficient combination reported absorbed nearly 70 percent. Mineral fertilizer absorption is typically somewhat above 50 percent in Europe, while the rates for Asian rice are 30 to 35 percent (Smil, 1999).

The rest of the N is lost. Most of the N losses are not directly emitted to the atmosphere, but enter the N cascade through water. The share of losses originating from fertilized cropland is not easily identified. Smil (1999) attempted to derive a global estimate of N losses from fertilized cropland. He estimates that globally, in the mid-1990, some 37 million tonnes N were exported from cropland through nitrate leaching (17 million tonnes N) and soil erosion (20 tonnes N). In addition, a fraction of the volatilized ammonia from mineral fertilizer N (11 million tonnes N yr^{-1}) finally also reaches the surface waters after deposition (some 3 million tonnes N yr^{-1}).

This N is gradually denitrified in subsequent reservoirs of the nitrogen cascade (Galloway *et al.*, 2003). The resulting enrichment of aquatic ecosystems with reactive N results in emissions not only of N_2, but also nitrous oxide. Galloway *et al.* (2004) estimate the total anthropogenic N_2O emission from aquatic reservoirs to equal some 1.5 million tonnes N, originating from a total of some 59 million tonnes N transported to inland waters and coastal areas. Feed and forage production induces a loss of N to aquatic sources of some 8 to 10 million tonnes yr^{-1} if one assumes such losses to be in line with N-fertilization shares of feed and forage production (some 20-25 percent of the world total, see carbon section). Applying the overall rate of anthropogenic aquatic N_2O emissions (1.5/59) to the livestock induced mineral fertilizer N loss to aquatic reservoirs results in a livestock induced emissions from aquatic sources of around 0.2 million tonnes N N_2O.

3.3.3 Wasting of nitrogen in the livestock production chain

The efficiency of N assimilation by crops leaves much to be desired. To a large extent this low efficiency is owing to management factors, such as the often excessive quantity of fertilizers applied, as well as the form and timing of applications. Optimizing these parameters can result in an efficiency level as high as 70 percent. The remaining 30 percent can be viewed as inherent (unavoidable) loss.

The efficiency of N assimilation by livestock is even lower. There are two essential differences between N in animal production and N in crop N use the:

- overall assimilation efficiency is much lower; and
- wasting induced by non-optimal inputs is generally lower.

As a result the inherent N assimilation efficiency of animal products is low leading to high N wasting under all circumstances.

N enters livestock through feed. Animal feeds contain 10 to 40 grams of N per kilogram of dry matter. Various estimates show livestock's low

[10] Crop production, as defined by van der Hoek, includes pastures and grass. Reducing inputs and outputs of the N balance to reflect only the cropland balance (animal manure N down to 20 million tonnes N as in FAO/IFA, 2001; Smil, 1999, and removing the consumed grass N output) results in a crop product assimilation efficiency of 38 percent. Smil's definition of cropland N recovery rates is less broad, but it does include forage crops. Forage crops contain many leguminous species and, therefore, improve the overall efficiency. Removing them from the balance appears to have only a minor effect. Though, Smil expresses recovery as the N contained in the entire plant tissue. A substantial part of this is not harvested (he estimates crop residues to contain 25 million tonnes N): some of this is lost upon decomposition after crop harvest and some (14 million tonnes N) re-enters the following cropping cycle. Removing crop residues from the balance gives a harvested crop N recovery efficiency of 60/155 million tonnes N= 38 percent.

efficiency in assimilating N from feed. Aggregating all livestock species, Smil (1999) estimated that in the mid-1990s livestock excreted some 75 million tonnes N. Van der Hoek (1998) estimates that globally livestock products contained some 12 million tonnes N in 1994. These figures suggest an underlying assimilation efficiency of only 14 percent. Considering only crop-fed animal production, Smil (2002) calculated a similar average efficiency of 15 percent (33 million tonnes N from feed, forage and residues producing 5 million tonnes of animal food N). NRC (2003) estimated the United States livestock sector's N assimilation efficiency also at 15 percent (0.9 over 5.9 million tonne N). According to the IPCC (1997), the retention of nitrogen in animal products, i.e., milk, meat, wool and eggs, generally ranges from about 5 to 20 percent of the total nitrogen intake. This apparent homogeneity of estimations may well hide different causes such as low feed quality in semi-arid grazing systems and excessively N-rich diets in intensive systems.

Efficiency varies considerably between different animal species and products. According to estimates by Van der Hoek (1998) global N efficiency is around 20 percent for pigs and 34 percent for poultry. For the United States, Smil (2002) calculated the protein conversion efficiency of dairy products at 40 percent, while that of beef cattle is only 5 percent. The low N efficiency of cattle on a global scale is partly inherent, given they are large animals with long gestation periods and a high basal metabolic rate. But the global cattle herd also comprises a large draught animal population whose task is to provide energy, not protein. For example, a decade ago cattle and horses still accounted for 25 percent of China's agricultural energy consumption (Mengjie and Yi, 1996). In addition, in many areas of the world, grazing animals are fed at bare maintenance level, consuming without producing much.

As a result, a huge amount of N is returned to the environment through animal excretions. However, not all this excreted N is wasted. When used as organic fertilizer, or directly deposited on grassland or crop fields, some of the reactive N re-enters the crop production cycle. This is particularly the case for ruminants, therefore, their contribution to overall N loss to the environment is less than their contribution to N in animal waste. Smil (2002) also noted that "this (ruminant assimilation: ed.) inefficiency is irrelevant in broader N terms as long as the animals (ruminants: ed.) are totally grass-fed, or raised primarily on crop and food processing residues (ranging from straw to bran and from oilseed cakes to grapefruit rinds) that are indigestible or unpalatable for non-ruminant species. Such cattle feeding calls for no, or minimal – because some pastures are fertilized – additional inputs of fertilizer-N. Any society that would put a premium on reducing N losses in agro-ecosystems would thus produce only those two kinds of beef. In contrast, beef production has the greatest impact on overall N use when the animals are fed only concentrates, which are typically mixtures of cereal grains (mostly corn) and soybeans".

Significant emissions of greenhouse gases to the atmosphere do arise from losses of N from animal waste that contain large amounts of N and have a chemical composition which induces very high loss rates. For sheep and cattle, faecal excretion is usually about 8 grams of N per kilogram of dry matter consumed, regardless of the nitrogen content of the feed (Barrow and Lambourne, 1962). The remainder of the nitrogen is excreted in the urine, and as the nitrogen content of the diet increases, so does the proportion of nitrogen in the urine. In animal production systems where the animal intake of nitrogen is high, more than half of the nitrogen is excreted as urine.

Losses from manure occur at different stages: during storage; shortly after application or direct deposition to land and losses at later stages.

3.3.4 Nitrogen emissions from stored manure

During storage (including the preceding excretion in animal houses) the organically bound nitrogen in faeces and urine starts to mineralize to NH_3/NH_4^+, providing the substrate for nitrifiers and denitrifiers (and hence, eventual production of N_2O). For the most part, these excreted N compounds mineralize rapidly. In urine, typically over 70 percent of the nitrogen is present as urea (IPCC, 1997). Uric acid is the dominant nitrogen compound in poultry excretions. The hydrolysis of both urea and uric acid to NH_3/NH_4^+ is very rapid in urine patches.

Considering first N_2O emissions, generally only a very small portion of the total nitrogen excreted is converted to N_2O during handling and storage of managed waste. As stated above, the waste composition determines its potential mineralization rate, while the actual magnitude of N_2O emissions depend on environmental conditions. For N_2O emissions to occur, the waste must first be handled aerobically, allowing ammonia or organic nitrogen to be converted to nitrates and nitrites (nitrification). It must then be handled anaerobically, allowing the nitrates and nitrites to be reduced to N_2, with intermediate production of N_2O and nitric oxide (NO) (denitrification). These emissions are most likely to occur in dry waste-handling systems, which have aerobic conditions, and contain pockets of anaerobic conditions owing to saturation. For example, waste in dry lots is deposited on soil, where it is oxidized to nitrite and nitrate, and has the potential to encounter saturated conditions. There is an antagonism between emission risks of methane versus nitrous oxide for the different waste storage pathways – trying to reduce methane emissions may well increase those of N_2O.

The amount of N_2O released during storage and treatment of animal wastes depends on the system and duration of waste management and the temperature. Unfortunately, there is not enough quantitative data to establish a relationship between the degree of aeration and N_2O emission from slurry during storage and treatment. Moreover, there is a wide range of estimates for the losses. When expressed in N_2O N/kg nitrogen in the waste (i.e. the share of N in waste emitted to the atmosphere as nitrous oxide), losses from animal waste during storage range from less than 0.0001 kg N_2O N/kg N for slurries to more than 0.15 kg N_2O N/kg nitrogen in the pig waste of deep-litter stables. Any estimation of global manure emission needs to consider these uncertainties. Expert judgement, based on existing manure management in different systems and world regions, combined with default IPCC emission factors (Box 3.3),[11] suggests N_2O emissions from stored manure equivalent to 0.7 million tonnes N yr^{-1}.

Turning to ammonia, rapid degradation of urea and uric acid to ammonium leads to very significant N losses through volatilization during storage and treatment of manure. While actual emissions are subject to many factors, particularly the manure management system and ambient temperature, most of the NH_3 N volatilizes during storage (typically about one-third of initially voided N), and before application or discharge. Smil (1999) (Galloway *et al.*, 2003 used Smil's paper for estimate) estimate that globally about 10 million tonnes of NH_3 N were lost to the atmosphere from confined animal feeding operations in the mid 1990s. Although, only a part of all collected manure originates from industrial systems.

On the basis of the animal population in industrial systems (Chapter 2), and their estimated manure production (IPCC, 1997), the current amount of N in the corresponding animal waste can be estimated at 10 million tonnes, and the corresponding NH_3 volatilization from stored manure at 2 million tonnes N.

Thus, volatilization losses during animal waste

[11] See also Annex 3.3. Regional livestock experts provided information on the relative importance of different waste management systems in each of the region's production systems through a questionnaire. On the basis of this information, waste management and gaseous emission experts from the Recycling of Agricultural, Municipal and Industrial Residues in Agriculture Network (RAMIRAN; available at www.ramiran.net) estimated region and system specific emissions.

management are not far from those from current synthetic N fertilizer use. On the one hand, this nitrogen loss reduces emissions from manure once applied to fields; on the other, it gives rise to nitrous oxide emissions further down the "nitrogen cascade."

3.3.5 Nitrogen emissions from applied or deposited manure

Excreta freshly deposited on land (either applied by mechanical spreading or direct deposition by the livestock) have high nitrogen loss rates, resulting in substantial ammonia volatilization. Wide variations in the quality of forages consumed by ruminants and in environmental conditions make N emissions from manure on pastures difficult to quantify. FAO/IFA (2001) estimate the N loss via NH_3 volatilization from animal manure, after application, to be 23 percent worldwide. Smil (1999) estimates this loss to be at least 15–20 percent.

The IPCC proposes a standard N loss fraction from ammonia volatilization of 20 percent, without differentiating between applied and directly deposited manure. Considering the substantial N loss from volatilization during storage (see preceding section) the total ammonia volatilization following excretion can be estimated at around 40 percent. It seems reasonable to apply this rate to directly deposited manure (maximum of 60 percent or even 70 percent have been recorded), supposing that the lower share of N in urine in tropical land-based systems is compensated by the higher temperature. We estimate that in the mid-1990s around 30 million tonnes of N was directly deposited on land by animals in the more extensive systems, producing an NH_3 volatilization loss of some 12 million tonnes N.[12]

[12] From the estimated total of 75 million tonnes N excreted by livestock we deduce that 33 million tonnes were applied to intensively used grassland, upland crops and wetland rice (FAO/IFA, 2001) and there were 10 million tonnes of ammonia losses during storage. Use of animal manure as fuel is ignored.

Added to this, according to FAO/IFA (2001) the post application loss of managed animal manure was about 8 million tonnes N, resulting in a total ammonia volatilization N loss from animal manure on land of around 20 million tonnes N.

These figures have increased over the past decade. Even following the very conservative IPCC ammonia volatilization loss fraction of 20 percent and subtracting manure used as for fuel results in an estimated NH_3 volatilization loss following manure application/deposition of some 25 million tonnes N in 2004.

Turning now to N_2O, the soil emissions originating from the remaining external nitrogen input (after subtraction of ammonia volatilization) depend on a variety of factors, particularly soil water filled pore space, organic carbon availability, pH, soil temperature, plant/crop uptake rate and rainfall characteristics (Mosier *et al.*, 2004). However, because of the complex interaction and the highly uncertain resulting N_2O flux, the revised IPCC guidelines are based on N inputs only, and do not consider soil characteristics. Despite this uncertainty, manure-induced soil emissions are clearly the largest livestock source of N_2O worldwide. Emission fluxes from animal grazing (unmanaged waste, direct emission) and from the use of animal waste as fertilizer on cropland are of a comparable magnitude. The grazing-derived N_2O emissions are in the range of 0.002–0.098 kg N_2O–N/kg nitrogen excreted, whereas the default emission factor used for fertilizer use is set at 0.0125 kg N_2O–N/kg nitrogen. Nearly all data pertain to temperate areas and to intensively managed grasslands. Here, the nitrogen content of dung, and especially urine, are higher than from less intensively managed grasslands in the tropics or subtropics. It is not known to what extent this compensates for the enhanced emissions in the more phosphorus-limited tropical ecosystems.

Emissions from applied manure must be calculated separately from emissions from waste excreted by animals. The FAO/IFA study (2001) estimates the N_2O loss rate from applied manure

Box 3.3 A new assessment of nitrous oxide emissions from manure by production system, species and region

The global figures we have cited demonstrate the importance of nitrous oxide emissions from animal production. However, to set priorities in addressing the problem, we need a more detailed understanding of the origin of these emissions, by evaluating the contribution of different production systems, species and world regions to the global totals.

Our assessment, detailed below, is based on current livestock data and results in a much higher estimate than most recent literature, which is based on data from the mid-1990s. The livestock sector has evolved substantially over the last decade. We estimate a global N excretion of some 135 million tonnes per year, whereas recent literature (e.g. Galloway *et al.*, 2003) still cites an estimate of 75 million tonnes yr^{-1} derived from mid-1990s data.

Our estimates of N_2O emissions from manure and soils are the result of combining current livestock production and population data (Groenewold, 2005) with the IPCC methodology (IPCC, 1997). Deriving N_2O emissions from manure management requires a knowledge of:

- N excretion by livestock type,
- the fraction of manure handled in each of the different manure management systems, and
- an emission factor (per kg N excreted) for each of the manure management systems.

The results are summed for each livestock species within a world region/production system (see Chapter 2) and multiplied by N excretion for that livestock type to derive the emission factor for N_2O per head.

Table 3.11

Estimated total N_2O emission from animal excreta in 2004

	N_2O emissions from manure management, after application/deposition on soil and direct emissions						
Region/country	**Dairy cattle**	**Other cattle**	**Buffalo**	**Sheep and goats**	**Pigs**	**Poultry**	**Total**
	(................ million tonnes per year ....)						
Sub-Saharan Africa	0.06	0.21	0.00	0.13	0.01	0.02	**0.43**
Asia excluding China and India	0.02	0.14	0.06	0.05	0.03	0.05	**0.36**
India	0.03	0.15	0.06	0.05	0.01	0.01	**0.32**
China	0.01	0.14	0.03	0.10	0.19	0.10	**0.58**
Central and South America	0.08	0.41	0.00	0.04	0.04	0.05	**0.61**
West Asia and North Africa	0.02	0.03	0.00	0.09	0.00	0.03	**0.17**
North America	0.03	0.20	0.00	0.00	0.04	0.04	**0.30**
Western Europe	0.06	0.14	0.00	0.07	0.07	0.03	**0.36**
Oceania and Japan	0.02	0.08	0.00	0.09	0.01	0.01	**0.21**
Eastern Europe and CIS	0.08	0.10	0.00	0.03	0.04	0.02	**0.28**
Other developed	0.00	0.03	0.00	0.02	0.00	0.00	**0.06**
Total	**0.41**	**1.64**	**0.17**	**0.68**	**0.44**	**0.36**	**3.69**
Livestock Production System							
Grazing	0.11	0.54	0.00	0.25	0.00	0.00	**0.90**
Mixed	0.30	1.02	0.17	0.43	0.33	0.27	**2.52**
Industrial	0.00	0.08	0.00	0.00	0.11	0.09	**0.27**

Source: Own calculations.

Box 3.3 **(cont.)**

Direct emissions resulting from manure applications (and grazing deposits) to soils were derived using the default emission factor for N applied to land (0.0125 kg N_2O-N/kg N). To estimate the amount of N applied to land, N excretion per livestock type was reduced allowing for the estimated fraction lost as ammonia and/or nitrogen oxides during housing and storage, the fraction deposited directly by grazing livestock, and the fraction used as fuel.

The results of these calculations (Table 3.11) show that emissions originating from animal manure are much higher than any other N_2O emissions caused by the livestock sector. In both extensive and intensive systems emissions from manure are dominated by soil emissions. Among soil emissions, emissions from manure management are more important. The influence of the characteristics of different production systems is rather limited. The strong domination of N_2O emissions by mixed livestock production systems is related in a rather linear way to the relative numbers of the corresponding animals. Large ruminants are responsible for about half the total N_2O emissions from manure.

Map 33 (Annex 1) presents the distribution among the world regions of the N_2O emissions of the different production systems.

at 0.6 percent,[13] i.e. lower than most mineral N fertilizers, resulting in an animal manure soil N_2O loss in the mid 1990s of 0.2 million tonnes N. Following the IPCC methodology would increase this to 0.3 million tonnes N.

Regarding animal waste excreted in pastures, dung containing approximately 30 million tonnes N was deposited on land in the more extensive systems in the mid-1990s. Applying the IPCC "overall reasonable average emission factor" (0.02 kg N_2O–N/kg of nitrogen excreted) to this total results in an animal manure soil N_2O loss of 0.6 million tonne N, making a total N_2O emission of about 0.9 million tonnes N in the mid-1990s.

Applying the IPCC methodology to the *current* estimate of livestock production system and animal numbers results in an overall "direct" animal manure soil N_2O loss totalling 1.7 million tonnes N per year. Of this, 0.6 million tonnes derive from grazing systems, 1.0 million tonnes from mixed and 0.1 million tonnes from industrial production systems (see Box 3.3).

3.3.6 Emissions following manure nitrogen losses after application and direct deposition

In the mid-1990s, after losses to the atmosphere during storage and following application and direct deposition, some 25 million tonnes of nitrogen from animal manure remained available per year for plant uptake in the world's croplands and intensively used grasslands. Uptake depends on the ground cover: legume/grass mixtures can take up large amount of added N, whereas loss from row crops[14] is generally substantial, and losses from bare/ploughed soil are much higher still.

If we suppose that N losses in grassland, through leaching and erosion, are negligible, and apply the crop N use efficiency of 40 percent to the remainder of animal manure N applied

[13] Expressed as a share of the initially applied amount, without deduction of the on-site ammonia volatilization, which may explain why the IPCC default is higher.

[14] Agricultural crops, such as corn and soybeans, that are grown in rows.

to cropland,[15] then we are left with some 9 or 10 million tonnes N that mostly entered the N cascade through water in the mid-1990s. Applying the N_2O loss rate for subsequent N_2O emission (Section 3.3.2) gives us an estimate of an additional emission of some 0.2 million tonne N N_2O from this channel. N_2O emissions of similar size can be expected to have resulted from the re-deposited fraction of the volatilized NH_3 from manure that reached the aquatic reservoirs in the mid-1990s.[16] Total N_2O emissions following N losses would, therefore, have been in the order of 0.30.4 million tonnes N N_2O per year in that period.

We have updated these figures for the current livestock production system estimates, using the IPCC methodology for indirect emissions. The current overall "indirect" animal manure N_2O emission following volatilization and leaching would then total around 1.3 million tonnes N per year. However, this methodology is beset with high uncertainties, and may lead to an overestimation because manure during grazing is considered. The majority of N_2O emissions, or about 0.9 million tonnes N, would still originate from mixed systems.

3.4 Summary of livestock's impact

Overall, livestock activities contribute an estimated 18 percent to total anthropogenic greenhouse gas emissions from the five major sectors for greenhouse gas reporting: energy, industry, waste, land use, land use change and forestry (LULUCF) and agriculture.

Considering the last two sectors only, livestock's share is over 50 percent. For the agriculture sector alone, livestock constitute nearly 80 percent of all emissions. Table 3.12 summarizes livestock's overall impact on climate change by: major gas, source and type of production system.

Here we will summarize the impact for the three major greenhouse gases.

Carbon dioxide

Livestock account for 9 percent of global anthropogenic emissions

When deforestation for pasture and feedcrop land, and pasture degradation are taken into account, livestock-related emissions of carbon dioxide are an important component of the global total (some 9 percent). However, as can be seen from the many assumptions made in preceding sections, these totals have a considerable degree of uncertainty. LULUCF sector emissions in particular are extremely difficult to quantify and the values reported to the UNFCCC for this sector are known to be of low reliability. This sector is therefore often omitted in emissions reporting, although its share is thought to be important.

Although small by comparison to LULUCF, the livestock food chain is becoming more fossil fuel intensive, which will increase carbon dioxide emissions from livestock production. As ruminant production (based on traditional local feed resources) shifts to intensive monogastrics (based on food transported over long distances), there is a corresponding shift away from solar energy harnessed by photosynthesis, to fossil fuels.

Methane

Livestock account for 35–40 percent of global anthropogenic emissions

The leading role of livestock, in methane emissions, has long been a well-established fact. Together, enteric fermentation and manure represent some 80 percent of agricultural methane emissions and about 35–40 percent of the total anthropogenic methane emissions.

With the decline of ruminant livestock in relative terms, and the overall trend towards higher productivity in ruminant production, it is unlikely

[15] FAO/IFA (2001) data on animal manure application to cropland, diminished by the FAO/IFA N volatilization and emission estimates.

[16] Applying the same N_2O loss rate for subsequent emission to the roughly 6 million tonnes N reaching the aquatic reservoirs out of the total of 22 million tonnes manure N volatilized as NH_3 in the mid-1990s according to the literature.

Table 3.12

Role of livestock in carbon dioxide, methane and nitrous oxide emissions

Gas	Source	Mainly related to extensive systems (10^9 tonnes CO_2 eq.)	Mainly related to intensive systems (10^9 tonnes CO_2 eq.)	Percentage contribution to total animal food GHG emissions
CO_2	**Total anthropogenic CO_2 emissions**	**24 (~31)**		
	Total from livestock activities	**~0.16 (~2.7)**		
	N fertilizer production		0.04	0.6
	on farm fossil fuel, feed		~0.06	0.8
	on farm fossil fuel, livestock-related		~0.03	0.4
	deforestation	(~1.7)	(~0.7)	34
	cultivated soils, tillage		(~0.02)	0.3
	cultivated soils, liming		(~0.01)	0.1
	desertification of pasture	(~0.1)		1.4
	processing		0.01 – 0.05	0.4
	transport		~0.001	
CH_4	**Total anthropogenic CH_4 emissions**	**5.9**		
	Total from livestock activities	**2.2**		
	enteric fermentation	1.6	0.20	25
	manure management	0.17	0.20	5.2
N_2O	**Total anthropogenic N_2O emissions**	**3.4**		
	Total from livestock activities	**2.2**		
	N fertilizer application		~0.1	1.4
	indirect fertilizer emission		~0.1	1.4
	leguminous feed cropping		~0.2	2.8
	manure management	0.24	0.09	4.6
	manure application/deposition	0.67	0.17	12
	indirect manure emission	~0.48	~0.14	8.7
Grand total of anthropogenic emissions		**33 (~40)**		
Total emissions from livestock activities		**~4.6 (~7.1)**		
Total extensive vs. intensive livestock system emissions		**3.2 (~5.0)**	**1.4 (~2.1)**	
Percentage of total anthropogenic emissions		10 (~13%)	4 (~5%)	

Note: All values are expressed in billion tonnes of CO_2 equivalent; values between brackets are or include emission from the land use, land-use change and forestry category; relatively imprecise estimates are preceded by a tilde.
Global totals from CAIT, WRI, accessed 02/06. Only CO_2, CH_4 and N_2O emissions are considered in the total greenhouse gas emission.
Based on the analyses in this chapter, livestock emissions are attributed to the sides of the production system continuum (from extensive to intensive/industrial) from which they originate.

that the importance of enteric fermentation will increase further. However, methane emissions from animal manure, although much lower in absolute terms, are considerable and growing rapidly.

Nitrous oxide

Livestock account for 65 percent of global anthropogenic emissions

Livestock activities contribute substantially to the emission of nitrous oxide, the most potent of the three major greenhouse gases. They contribute almost two-thirds of all anthropogenic N_2O emissions, and 75–80 percent of agricultural emissions. Current trends suggest that this level will substantially increase over the coming decades.

Ammonia

Livestock account for 64 percent of global anthropogenic emissions

Global anthropogenic atmospheric emission of ammonia has recently been estimated at some 47 million tonnes N (Galloway *et al.*, 2004). Some 94 percent of this is produced by the agricultural sector. The livestock sector contributes about 68 percent of the agriculture share, mainly from deposited and applied manure.

The resulting air and environmental pollution (mainly eutrophication, also odour) is more a local or regional environmental problem than a global one. Indeed, similar levels of N depositions can have substantially different environmental effects depending on the type of ecosystem they affect. The modelled distribution of atmospheric N deposition levels (Figure 3.3) are a better indication of the environmental impact than the global figures. The distribution shows a strong and clear co-incidence with intensive livestock production areas (compare with Map 13).

The figures presented are estimates for the overall global-level greenhouse gas emissions. However, they do not describe the entire issue of livestock-induced change. To assist decision-making, the level and nature of emissions need to be understood in a local context. In Brazil, for example, carbon dioxide emissions from land-use change (forest conversion and soil organic matter loss) are reported to be much higher than emissions from the energy sector. At the same time, methane emissions from enteric fermentation strongly dominate the country's total methane emission, owing to the extensive beef cattle population. For this same reason pasture soils produce the highest nitrous oxide emissions in Brazil, with an increasing contribution from manure. If livestock's role in land-use change is included, the contribution of the livestock sector to the total greenhouse gas emission of this very large country can be estimated to be as high as 60 percent, i.e. much higher than the 18 percent at world level (Table 3.12).

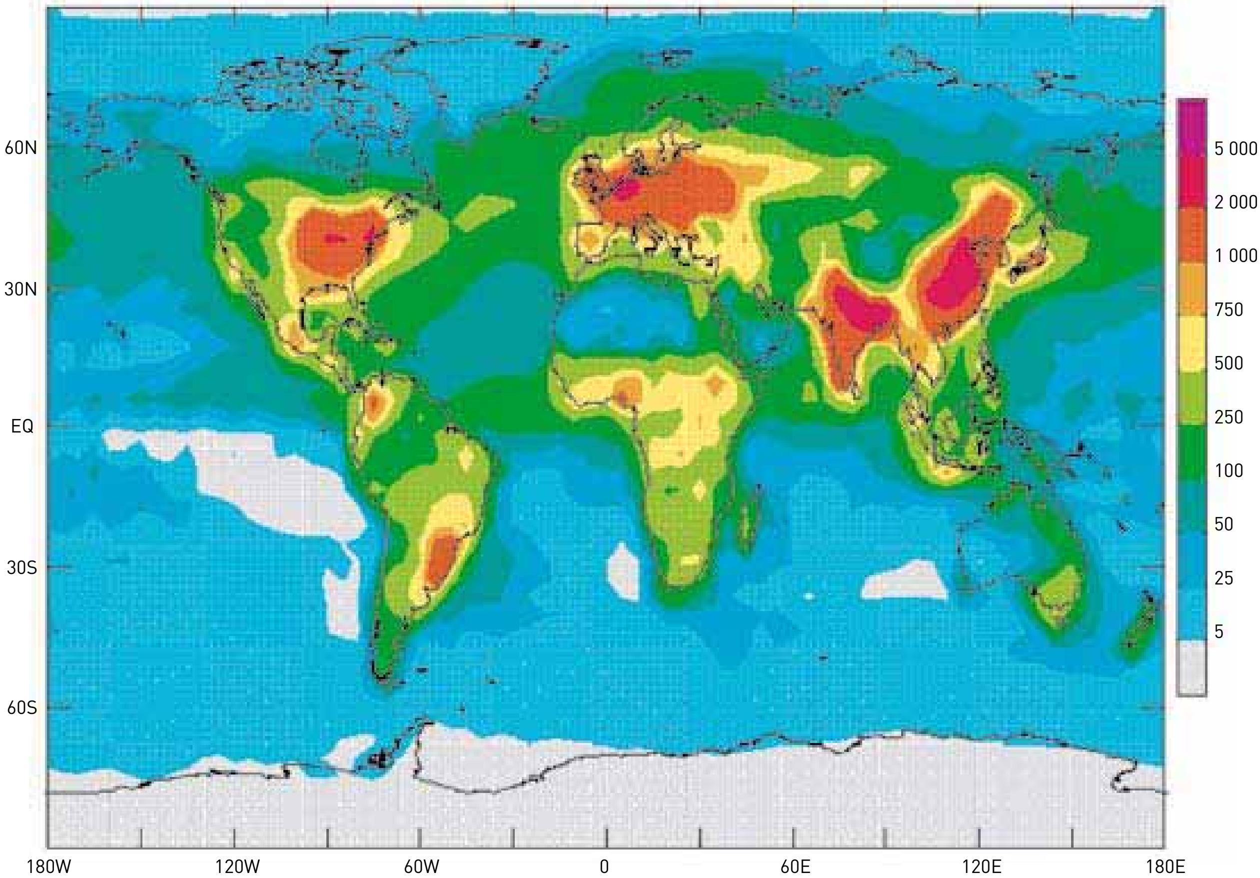
3.5 Mitigation options

Just as the livestock sector makes large and multiple contributions to climate change and air pollution, so there are multiple and effective options for mitigation. Much can be done, but to get beyond a "business as usual" scenario will require a strong involvement of public policy. Most of the options are not cost neutral – simply enhancing awareness will not lead to widespread adoption. Moreover, by far the largest share of emissions come from more extensive systems, where poor livestock holders often extract marginal livelihoods from dwindling resources and lack the funds to invest in change. Change is a matter of priority and vision, of making short-term expenses (for compensation or creation of alternatives) for long-term benefits.

We will examine the policy aspects in Chapter 6. Here we explore the main technical options, including those for substantially reducing the major current emissions and those that will create or expand substantial sinks.

Globally climate change is strongly associated with carbon dioxide emissions, which represent roughly three quarters of the total anthropogenic emissions. Because the energy sector accounts for about three-quarters of anthropogenic CO_2,

Figure 3.3 **Spatial pattern of total inorganic nitrogen deposition in the early 1990s**



Note: Units – mg N per square metre per year.
Source: Galloway *et al.*, (2004).

limited attention has been paid to reducing emissions of other gases from other sectors. In a development context, particularly, this is not justified. While developing countries account for only 36 percent of CO_2 emissions, they produce more than half of N_2O and nearly two-thirds of CH_4. It is therefore surprising to see that even in the case of a large country such as Brazil, most mitigation efforts focus on the energy sector.

3.5.1 Sequestering carbon and mitigating CO_2 emissions

Compared to the amounts of carbon released from changes in land use and land-degradation, emissions from the food chain are small. So for CO_2 the environmental focus needs to be on addressing issues of land-use change and land degradation. Here the livestock sector offers a significant potential for carbon sequestration, particularly in the form of improved pastures.

Reducing deforestation by agricultural intensification

When it comes to land-use change, the challenge lies in slowing and eventually halting and reversing deforestation. The still largely uncontrolled process urgently needs to be consciously planned, on the basis of trade-offs between benefits and costs at different spatial and temporal scales. Amazon deforestation, related to agricultural expansion for livestock, has been demonstrated to contribute substantially to global anthropogenic carbon dioxide emissions. The forecast increase in emissions could be curtailed if development strategies were implemented to control frontier expansion and create economic alternatives (Carvalho *et al.*, 2004).

Creating incentives for forest conservation and decreased deforestation, in Amazonia and other tropical areas, can offer a unique opportunity for climate change mitigation, especially given the ancillary benefits (see Chapter 6 on policies) and relative low costs. Any programme that aims to set aside land for the purpose of sequestering carbon must do so without threatening food security in the region. Vlek *et al.* (2004) consider that the only available option to free up the land necessary for carbon sequestration would be intensification of agricultural production on some of the better lands, for example by increased fertilizer inputs. They demonstrate that the increased carbon dioxide emissions related to the extra fertilizer production would be far outweighed by the sequestered or avoided emissions of organic carbon related to deforestation. Increased fertilizer use though constitutes just one of many options for intensification. Others include higher-yielding, better adapted varieties and improved land and water management. Although rationally attractive, the "sequestration through intensification" paradigm may not be effective in all socio-political contexts and imposes strong conditions on the regulatory framework and its enforcement. Where deforestation occurs, and where it is accepted, care should be taken to quickly transform the area into a sustainable agricultural area, for example by implementing practices like silvo-pastoral systems (see Box 6.2, Chapter 6) and conservation agriculture, thus preventing irreversible damage.

Restoring soil organic carbon to cultivated soils

The relatively low carbon dioxide emissions from arable land leave little scope for significant mitigation. But there is a huge potential for net sequestration of carbon in cultivated soils. The carbon sink capacity of the world's agricultural and degraded soils is 50 to 66 percent of the historic carbon loss from soils of 42 to 78 gigatons of carbon (Lal, 2004a). In addition, carbon sequestration has the potential to enhance food security and to offset fossil fuel emissions.

Soil processes, with respect to carbon, are characterized by the dynamic equilibrium of input (photosynthesis) and output (respiration). Under conventional cultivation practices, the conversion of natural systems to cultivated agriculture results in losses of soil organic carbon (SOC) on the order of 20 to 50 percent of the pre-cultivation stocks in the top one metre (Paustian *et al.*, 1997; Lal and Bruce, 1999).

Changing environmental conditions and land management may induce a change in the equilibrium to a new level that is considered stable. There are now proven new practices that can improve soil quality and raise soil organic carbon levels. The full potential for terrestrial soil carbon sequestration is uncertain, because of insufficient data and understanding of SOC dynamics at all levels, including molecular, landscape, regional and global scales (Metting *et al.*, 1999). According to the IPCC (2000) improved practices typically allow soil carbon to increase at a rate of about 0.3 tonnes of carbon per hectare per year. If these practices were adopted on 60 percent of the available arable land worldwide, they would result in a capture of about 270 million tonnes C per year over the next few decades (Lal, 1997). It is unclear if this rate is sustainable: research shows a relatively rapid increase in carbon sequestration for a period of about 25 years and a gradual levelling thereafter (Lal *et al.*, 1998).

Non-conventional practices can be grouped into three classes: agricultural intensification, conservation tillage, and erosion reduction. Examples of intensifying practices are improved cultivars, irrigation, organic and inorganic fertilization, management of soil acidity, integrated pest management, double-cropping, and crop rotations including green manure and cover crops. Increasing crop yields result in more carbon accumulated in crop biomass or in an alteration of the harvest index. The higher crop residues, sometimes associated with higher yields, favour enhanced soil carbon storage (Paustian *et al.*, 1997).

IPCC (2000) provides an indication of the "carbon gain rate" that can be obtained for some practices.

Conservation tillage is any tillage and planting system in which 30 percent, or more, of the crop residue remains on the soil surface after planting. Generally it also comprises reduced mechanical intervention during the cropping season. Conservation tillage can include specific tillage types such as no-till, ridge-till, mulch-till, zone-till, and strip-till systems, chosen by farmers to address soil type, crop grown, machinery available, and local practice. Although these systems were originally developed to address problems of water quality, soil erosion and agricultural sustainability, they also lead to higher soil organic carbon and increased fuel efficiency (owing to reduced use of machinery for soil cultivation). Hence, at the same time, they increase carbon sinks and reduce carbon emissions.

Conservation tillage is achieving widespread adoption around the world. In 2001, a study conducted by the American Soybean Association (ASA) showed that a majority of the 500 000 soybean farmers in the United States had adopted conservation tillage practices following the introduction of herbicide-resistant soybeans (Nill, 2005). The resulting topsoil carbon increase also enables the land to absorb increasing amounts of rainfall, with a corresponding reduction in runoff and much better drought resistance compared to conventionally tilled soybeans.

The IPCC (2000) estimates that conservation tillage can sequester 0.1–1.3 tonnes C ha^{-1} y^{-1} globally, and could feasibly be adopted on up to 60 percent of arable lands. These benefits accrue only if conservation tillage continues: a return to intensive tillage or mould-board ploughing can negate or offset any gains and restore the sequestered carbon to the atmosphere. Soil carbon sequestration can be even further increased when cover crops are used in combination with conservation tillage.

Similar results have been reported from organic farming,[17] which has evolved since the early years of the twentieth century. Organic farming increases soil organic carbon content. Additional benefits are reported such as reversing of land degradation, increasing soil fertility and health. Trials of maize and soybean reported in Vasilikiotis (2001) demonstrated that organic systems can achieve yields comparable to conventional intensive systems, while also improving long-term soil fertility and drought resistance.

These improved agriculture practices are also the major components of sustainable agriculture and rural development as outlined in the UNCED Agenda 21 (Chapter 14). Although farmers' adoption of these practices also create on-farm benefits such as increased crop yields, the adoption of such practices on a wider scale largely depends on the extent that farmers are faced with the environmental consequences of their current practices. Farmers may also need additional knowledge and resources before they will invest in such practices. Farmers will make their own choices, depending on expected net returns, in the context of existing agriculture and environmental policies.

[17] Organic farming is the outcome of theory and practice since the early years of the twentieth century, involving a variety of alternative methods of agricultural production mainly in northern Europe. There have been three important movements: biodynamic agriculture, which appeared in Germany; organic farming, which originated in England; and biological agriculture, which was developed in Switzerland. Despite some differences of emphasis, the common feature of all these movements is to stress the essential link between farming and nature, and to promote respect for natural equilibria. They distance themselves from the conventional approach to farming, which maximizes yields through the use of various kinds of synthetic products.

Table 3.13

Global terrestrial carbon sequestration potential from improved management

Carbon sink	Potential sequestration (billion tonnes C per year)
Arable lands	0.85 – 0.90
Biomass crops for biofuel	0.5 – 0.8
Grassland and rangelands	1.7
Forests	1–2

Source: adapted from Rice (1999).

Reversing soil organic carbon losses from degraded pastures

Up to 71 percent of the world's grasslands were reported to be degraded to some extent in 1991 (Dregne *et al*. 1991) as a result of overgrazing, salinization, alkalinization, acidification, and other processes.

Improved grassland management is another major area where soil carbon losses can be reversed leading to net sequestration, by the use of trees, improved species, fertilization and other measures. Since pasture is the largest anthropogenic land use, improved pasture management could potentially sequester more carbon than any other practice (Table 4-1, IPCC, 2000). There would also be additional benefits, particularly preserving or restoring biodiversity. It can yield these benefits in many ecosystems.

In the humid tropics silvo-pastoral systems (discussed in Chapter 6, Box 6.2) are one approach to carbon sequestration and pasture improvement.

In dryland pastures soils are prone to degradation and desertification, which have lead to dramatic reductions in the SOC pool (see Section 3.2.1 on *livestock-related emissions from cultivated soils*) (Dregne, 2002). However, some aspects of dryland soils may help in carbon sequestration. Dry soils are less likely to lose carbon than wet soils, as lack of water limits soil mineralization and therefore the flux of carbon to the atmosphere. Consequently, the residence time of carbon in dryland soils is sometimes even longer than in forest soils. Although the rate at which carbon can be sequestered in these regions is low, it may be cost-effective, particularly taking into account all the side-benefits for soil improvement and restoration (FAO, 2004b). Soil-quality improvement as a consequence of increased soil carbon will have an important social and economic impact on the livelihood of people living in these areas. Moreover, there is a great potential for carbon sequestration in dry lands because of their large extent and because substantial historic carbon losses mean that dryland soils are now far from saturation.

Some 18–28 billion tonnes of carbon have been lost as a result of desertification (see section on feed sourcing). Assuming that two-thirds of this can be re-sequestered through soil and vegetation restoration (IPCC, 1996), the potential of C sequestration through desertification control and restoration of soils is 12–18 billion tonnes C over a 50 year period (Lal, 2001, 2004b). Lal (2004b) estimates that the "eco-technological" (maximum achievable) scope for soil carbon sequestration in the dryland ecosystems may be about 1 billion tonnes C yr^{-1}, though he suggests that realization of this potential would require a "vigorous and a coordinated effort at a global scale towards desertification control, restoration of degraded ecosystems, conversion to appropriate land uses, and adoption of recommended management practices on cropland and grazing land." Taking just the grasslands in Africa, if the gains in soil carbon stocks, technologically achievable with improved management, were actually achieved on only 10 percent of the area concerned, this would result in a SOC gain rate of 1 328 million tonnes C per year for some 25 years (Batjes, 2004). For Australian rangelands, which occupy 70 percent of the country's land mass, the potential sequestration rate through better management has been evaluated at 70 million tonnes C per year (Baker *et al*., 2000).

Overgrazing is the greatest cause of degradation of grasslands and the overriding human-influenced factor in determining their soil carbon levels. Consequently, in many systems, improved grazing management, such as optimizing stock numbers and rotational grazing, will result in substantial increases in carbon pools (Table 4–6, IPCC, 2000).

Many other technical options exist, including fire management, protection of land, set-asides and grassland production enhancement (e.g., fertilization, introduction of deep-rooted and legume species). Models exist to provide an indication of the respective effects of these practices in a particular situation. More severely degraded land requires landscape rehabilitation and erosion control. This is more difficult and costly, but Australian research reports considerable success in rehabilitating landscape function by promoting the rebuilding of patches (Baker *et al.*, 2000).

Because dryland conditions offer few economic incentives to invest in land rehabilitation for agricultural production purposes, compensation schemes for carbon sequestration may be necessary to tip the balance in some situations. A number of mechanisms stimulated by the UNFCCC are now operational (see Chapter 6). Their potential may be high in pastoral dry lands, where each household ranges livestock over large areas. Typical population densities in pastoral areas are 10 people per km^2 or 1 person per 10 ha. If carbon is valued at US$10 per tonne and modest improvements in management can gain 0.5 tonnes C/ha/yr, individuals might earn US$50 a year for sequestering carbon. About half of the pastoralists in Africa earn less than US$1 per day or about US$360 per year. Thus, modest changes in management could augment individual incomes by 15 percent, a substantial improvement (Reid *et al.*, 2004). Carbon improvements might also be associated with increases in production, creating a double benefit.

Carbon sequestration through agroforestry

In many situations agroforestry practices also offer excellent, and economically viable, potential for rehabilitation of degraded lands and for carbon sequestration (IPCC, 2000; FAO, 2000). Despite the higher carbon gains that might come from agroforestry, Reid *et al.* (2004) estimate that the returns per person are likely to be lower in these systems because they principally occur in higher-potential pastoral lands, where human population densities are 3–10 times higher than in drier pastoral lands. Payment schemes for carbon sequestration through silvo-pastoral systems have already proven their viability in Latin American countries (see Box 6.2, Chapter 6).

Unlocking the potential of mechanisms like carbon credit schemes is still a remote goal, not only requiring vigorous and coordinated effort on a global scale, but also the overcoming of a number of local obstacles. As illustrated by Reid *et al.* (2004), carbon credit schemes will require communication between groups often distant from one another, yet pastoral areas usually have less infrastructure and much lower population density than higher potential areas. Cultural values may pose constraints but sometimes offer opportunities in pastoral lands. Finally the strength and ability of government institutions required to implement such schemes is often insufficient in the countries and areas where they are most needed.

3.5.2 Reducing CH_4 emissions from enteric fermentation through improved efficiency and diets

Methane emissions by ruminants are not only an environmental hazard but also a loss of productivity, since methane represents a loss of carbon from the rumen and therefore an unproductive use of dietary energy (US-EPA, 2005). Emissions per animal and per unit of product are higher when the diet is poor.

The most promising approach for reducing methane emissions from livestock is by improving the productivity and efficiency of livestock production, through better nutrition and genetics. Greater efficiency means that a larger portion of the energy in the animals' feed is directed toward the creation of useful products (milk, meat, draught power), so that methane emissions per unit product are reduced. The trend towards high performing animals and towards monogastrics and poultry in particular, are valuable in this context as they reduce methane per unit of product. The increase in production efficiency also leads to a reduction in the size of the herd required to produce a given level of product. Because many developing countries are striving to increase production from ruminant animals (primarily milk and meat), improvements in production efficiency are urgently needed for these goals to be realized without increasing herd sizes and corresponding methane emissions.

A number of technologies exist to reduce methane release from enteric fermentation. The basic principle is to increase the digestibility of feedstuff, either by modifying feed or by manipulating the digestive process. Most ruminants in developing countries, particularly in Africa and South Asia, live on a very fibrous diet. Technically, the improvement of these diets is relatively easy to achieve through the use of feed additives or supplements. However, such techniques are often difficult to adopt for smallholder livestock producers who may lack the necessary capital and knowledge.

In many instances, such improvements may not be economical, for example where there is insufficient demand or infrastructure. Even in a country like Australia, low-cost dairy production focuses on productivity per hectare rather than per cow, so many options for reducing emissions are unattractive – e.g. dietary fat supplementation or increased grain feeding (Eckard *et al*., 2000). Another technical option is to increase the level of starch or rapidly fermentable carbohydrates in the diet, so as to reduce excess hydrogen and subsequent CH_4 formation. Again low-cost extensive systems may not find it viable to adopt such measures. However, national planning strategies in large countries could potentially bring about such changes. For example, as Eckard *et al*. (2000) suggest, concentrating dairy production in the temperate zones of Australia could potentially decrease methane emissions, because temperate pastures are likely to be higher in soluble carbohydrates and easily digestible cell wall components.

For the United States, US-EPA (2005) reports that greater efficiency of livestock production has already led to an increase in milk production while methane emissions decreased over the last several decades. The potential for efficiency gains (and therefore for methane reductions) is even larger for beef and other ruminant meat production, which is typically based on poorer management, including inferior diets. US-EPA (2005) lists a series of management measures that could improve a livestock operation's production efficiency and reduce greenhouse gas emissions, including:

- improving grazing management;
- soil testing, followed by addition of proper amendments and fertilizers;
- supplementing cattle diets with needed nutrients;
- developing a preventive herd health programme;
- providing appropriate water sources and protecting water quality; and
- improving genetics and reproductive efficiency.

When evaluating techniques for emission reduction it is important to recognize that feed and feed supplements used to enhance productivity may well involve considerable greenhouse gas emissions to produce them, which will affect the balance negatively. If production of such feed stuffs is to increase substantially, options to reduce emissions at feed production level will also need to be considered.

More advanced technologies are also being studied, though they are not yet operational. These include:

- reduction of hydrogen production by stimulating acetogenic bacteria;
- defaunation (eliminating certain protozoa from the rumen); and
- vaccination (to reduce methanogens).

These options would have the advantage of being applicable to free-ranging ruminants as well, although the latter option may encounter resistance from consumers (Monteny *et al.*, 2006). Defaunation has been proven to lead to a 20 percent reduction in methane emissions on average (Hegarty, 1998), but regular dosing with the defaunating agent remains a challenge.

3.5.3 Mitigating CH_4 emissions through improved manure management and biogas

Methane emissions from anaerobic manure management can be readily reduced with existing technologies. Such emissions originate from intensive mixed and industrial systems; these commercially oriented holdings usually have the capacity to invest in such technologies.

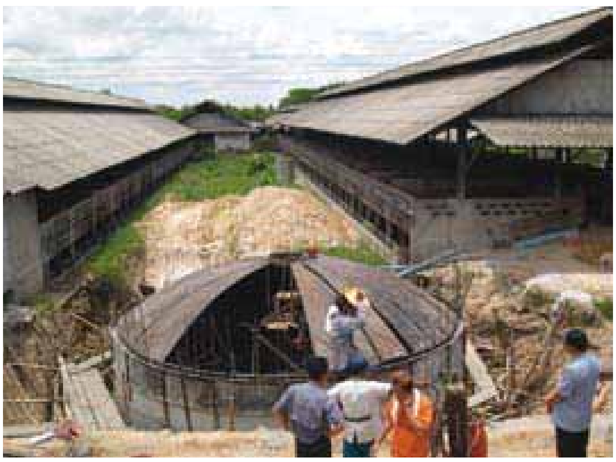
The potential for emission abatement from manure management is considerable and multiple options exist. A first obvious option to consider is balanced feeding, as it also influences other emissions. Lower carbon to nitrogen ratios in feed lead to increased methane emissions, in an exponential fashion. Manure with high nitrogen content will emit greater levels of methane than manure with lower N contents. Hence increasing the C to N ratio in feeds can reduce emissions.

The temperature at which manure is stored can significantly affect methane production. In farming systems where manure is stored in the stabling (e.g. in pig farms where effluents are stored in a pit in the cellar of a stable) emissions can be higher than when manure is stored outside at lower ambient temperatures. Frequent and complete removal of the manure from the indoor storage pits reduces methane emissions effectively in temperate climates, but only where there is sufficient outdoor storage capacity (and additional measures to prevent CH_4 emissions outdoors). Reduction of gas production can also be achieved through deep cooling of manure (to below 10°C), though this requires higher investment and also energy consumption with a risk of increased carbon dioxide emissions. Cooling of pig slurry can reduce in-house CH_4 (and N_2O) emissions by 21 percent relative to not cooling (Sommer *et al.*, 2004).

Additional measures include anaerobic digestion (producing biogas as an extra benefit), flaring/burning (chemical oxidation; burning), special biofilters (biological oxidation) (Monteny *et al.*, 2006; Melse and van der Werf, 2005), composting and aerobic treatment. Biogas is produced by controlled anaerobic digestion – the bacterial fermentation of organic material under controlled conditions in a closed vessel. Biogas is typically made up of 65 percent methane and 35 percent carbon dioxide. This gas can be burned directly for heating or light, or in modified gas boilers to run internal combustion engines or generators.

It is assumed that biogas can achieve a 50 percent reduction in emissions in cool climates for manures which would otherwise be stored as liquid slurry (and hence have relatively high methane emissions). For warmer climates, where methane emissions from liquid slurry manure storage systems are estimated to be over three times higher (IPCC, 1997), a reduction potential of 75 percent is possible (Martinez, personal communication).

Various systems exist to exploit this huge potential, such as covered lagoons, pits, tanks and other liquid storage structures. These would be suitable for large or small-scale biogas sys-



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Anaerobic digestor for biogas production in a commercial pig farm – Central Thailand 2005 tems, with a wide range of technological options and different degrees of sophistication. Additionally, covered lagoons and biogas systems produce a slurry that can be applied to rice fields instead of untreated dung, leading to reduced methane emissions (Mendis and Openshaw, 2004). These systems are common practice in much of Asia, particularly in China. In Vietnam, Thailand and the Philippines biogas is also widely used. A new opportunity in hot climate is the use of biogas to fuel modern cooling systems (e.g. EVAP system) and thereby achieve substantial savings on energy costs.

However, in most of these countries biogas has been helped to spread by subsidy schemes or other forms of promotion. Current uptake of biogas technologies is limited in many countries because of insufficient regulatory frameworks and absence of appropriate financial incentives. The wider use of biogas systems (for use on-farm or for delivering electricity to the public net) depends on the relative price of other energy sources. Usually biogas systems are not competitive in the absence of subsidies, other than in remote locations where electricity and other forms of energy are unavailable or unreliable. Biogas feasibility also depends on the degree to which there are options to co-digest waste products so as to increase gas production (see Nielsen and Hjort-Gregersen, 2005).

The further development and promotion of controlled anaerobic digestion will have substantial additional positive effects related to other environmental problems caused by animal wastes, and/or the promotion of renewable energy sources. For example, anaerobic digestion offers benefits in terms of reduced odour and pathogens.

Although more time consuming for the farmer, possible solutions to reduce methane emissions also lie in shifting towards solid manure management. Aerobic treatments can also be used to reduce methane emissions and odour. In practice they are applied to liquid manures through aeration and to solid manures by composting and often have a positive side-effect on pathogen content.

3.5.4 Technical options for mitigating N_2O emissions and NH_3 volatilization

The best way to manage the continuing human interference in the nitrogen cycle is to maximize the efficiencies of human uses of N (Smil, 1999).

Reducing the nitrogen content of manures as suggested above may also lead to lower N_2O emissions from stables, during storage, and after application to soil.

An important mitigation pathway lies in raising the low animal nitrogen assimilation efficiency (14 percent, against some 50 percent for crops – see Sections 3.3.2 and 3.3.3) through more balanced feeding (i.e. by optimizing proteins or amino acids to match the exact requirements of individual animals or animal groups). Improved feeding practices also include grouping animals by gender and phase of production, and improving the feed conversion ratio by tailoring feed to physiological requirements. However, even when good management practices are used to minimize nitrogen excretion, large quantities still remain in the manure.

Another possible intervention point is immediately after reactive nitrogen is used as a resource (e.g. digestion of feed), but before it is distributed to the environment. In intensive production, substantial N losses can occur during storage primarily through volatilization of ammonia. The use of an enclosed tank can nearly eliminate this loss. Maintaining a natural crust on the manure surface in an open tank is almost as effective and more economical. However, the first option offers an important potential synergy with respect to mitigating methane emissions.

N_2O emissions from slurry applications to grassland were reduced when slurry was stored for 6 months or passed through an anaerobic digester prior to spreading (Amon *et al.*, 2002). It can be inferred that during storage and anaerobic digestion readily available C (which would otherwise fuel denitrification and increase gaseous N loss) is incorporated into microbial biomass or lost as CO_2 or CH_4. Hence there is less available C in the slurry to fuel denitrification when the slurry is applied to land. It follows that anaerobic digestion, e.g. for biogas production, can substantially mitigate nitrous oxide and methane emissions (provided the biogas is used and not discharged). In addition, electricity can be generated and N_2O emissions from the spread of (digested) slurry would also be reduced.

The identification and choice of other N_2O emission mitigation options during storage are complex, and the choice is also limited by farm and environmental constraints and costs. Important trade-offs exist between methane and nitrous oxide emissions: technologies with potential to reduce nitrous oxide emissions often increase those of methane and vice versa. A management change from straw- to slurry-based systems for example may result in lower N_2O emission, but increased CH_4 emission. Also, compaction of solid manure heaps to reduce oxygen entering the heap and maintaining anaerobic conditions has had mixed success in reducing N_2O emissions (Monteny *et al.*, 2006), and may increase CH_4 emissions.

Much of the challenge of reducing emissions of NH_3 and N_2O falls upon crop farmers. Rapid incorporation and shallow injection methods for manure reduce N loss to the atmosphere by at least 50 percent, while deep injection into the soil essentially eliminates this loss (Rotz, 2004) (losses via leaching may increase though). Use of a crop rotation that can efficiently recycle nutrients, and applying N near the time it is needed by crops reduces the potential for further losses. In more generic terms, the key to reducing nitrous oxide emissions is the fine-tuning of waste application to land with regard to environmental conditions, including timing, amounts and form of application in response to crop physiology and climate.

Another technological option for reducing emissions during the application/deposition phase is the use of nitrification inhibitors (NIs) that can be added to urea or ammonium compounds. Monteny *et al.* (2006) cite examples of substantially reduced emissions. Some of these substances can potentially be used on pastures where they act upon urinary N, an approach being adopted in New Zealand (Di and Cameron, 2003). Costs of NIs may be offset by increased crop/pasture N uptake efficiency. The degree of adoption of NIs may depend on public perception of introducing yet another chemical into the environment (Monteny *et al.*, 2006).

Options to reduce emissions from grazing systems are particularly important as they constitute the bulk of nitrous oxide emissions. For grazing animals, excessive losses from manure can be avoided by not overstocking pastures and avoiding late fall and winter grazing.

Finally, land drainage is another option to reduce nitrous oxide emissions before N enters the next phase of the nitrogen cascade. Improvement of soil physical conditions to reduce soil wetness in the more humid environments, and especially in grassland systems, may significantly reduce N_2O emissions. Soil compaction by traffic, tillage and grazing livestock can increase the anaerobicity of the soil and enhance conditions for denitrification.

This section covered the technical options that have the largest mitigation potential and are of global interest. Many other options could be presented and their potential analyzed,[18] but mostly the latter would be far less significant and their applicability to different systems and regions not as wide. Among the selection of options presented, those that contribute to the mitigation of several gases at a time (anaerobic digestion of manure), as well as those that provide other environmental benefits in parallel (e.g. pasture management) deserve special attention.

[18] Mitigation options that more specifically focus on limiting nitrate losses to water, though also relevant here, are presented in the following chapter.