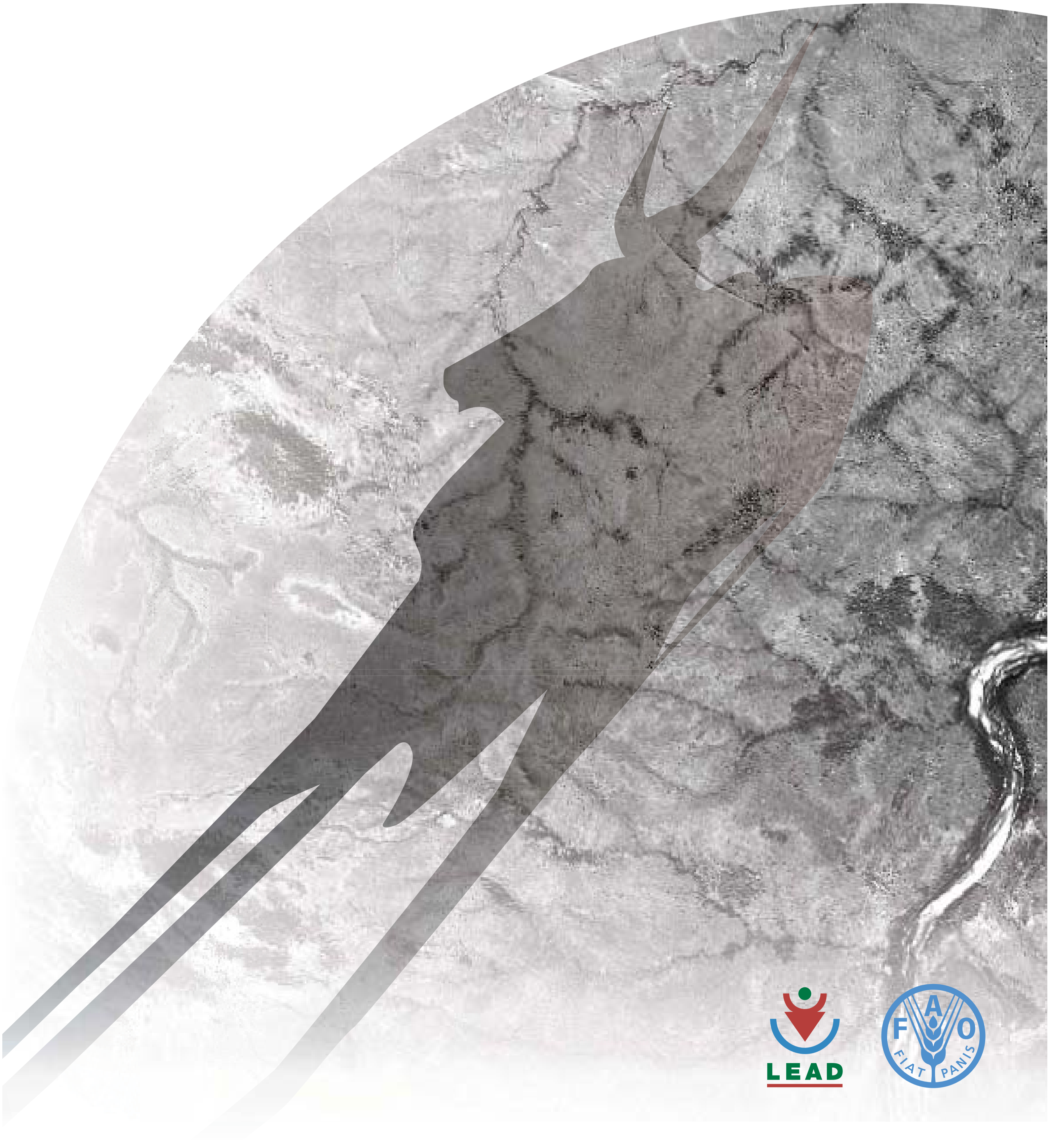
livestock's long shadow

environmental issues and options



LEAD

FAO FIAT PANIS

The Livestock, Environment and Development (LEAD) Initiative is supported by the World Bank, the European Union (EU), the Ministère des Affaires Etrangères (France), German Federal Ministry for Economic Cooperation and Development via GTZ (Germany), the Department for International Development (United Kingdom), the US Agency for International Development (USA), the International Development Agency (Denmark), the Swiss Agency for Development and Cooperation (Switzerland), the International Fund for Agricultural Development (IFAD) and the Food and Agriculture Organization of the United Nations (FAO).

LEAD website: http://www.virtualcentre.org

Photographs

Page 2 and 3: USDA / Ken Hammond
Page 22 and 23: USDA / Dana Downic
Page 78 and 79: FAO / R. Faidutti
Page 124 and 125: USDA–NRCS / Gene Alexander
Page 180 and 181: CIPAV – Photograhic archive
Page 220 and 221: FAO / H. Wagner
Page 266 and 267: Nikolaus Schareika

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS

Rome, 2006

livestock's long shadow

environmental issues and options

Henning Steinfeld
Pierre Gerber
Tom Wassenaar
Vincent Castel
Mauricio Rosales
Cees de Haan

ISBN 978-92-5-105571-7

Preface

The in-depth assessment presented in this document of the various significant impacts of the world's livestock sector on the environment is deliberately termed *Livestock's long shadow* so as to help raise the attention of both the technical and the general public to the very substantial contribution of animal agriculture to climate change and air pollution, to land, soil and water degradation and to the reduction of biodiversity. This is not done simply to blame the rapidly growing and intensifying global livestock sector for severely damaging the environment but to encourage decisive measures at the technical and political levels for mitigating such damage. The detailed assessment of the various environmental impacts of the sector is therefore associated with the outline of technical and policy-related action to address these impacts.

The assessment builds on the work of the Livestock, Environment and Development (LEAD) Initiative. This multi-stakeholder Initiative, coordinated by FAO's Animal Production and Health Division, was formed to address the environmental consequences of livestock production, particularly in the light of rising demand for food products of animal origin and the increasing pressure on natural resources. The LEAD Initiative brought together a broad range of research and development institutions and individuals interested in livestock–environment interactions; it has been active in a number of areas of particular concern, i.e. in land and water pollution from intensive livestock production in land degradation from overgrazing in dry lands and in livestock-induced deforestation in the humid and subhumid tropics.

While previous assessments of the livestock–environment interactions by LEAD have adopted a livestock sector perspective, i.e. investigated the impacts of the sector on the natural resources used in animal production, the current assessment sets off from the environment and determines the contribution of livestock to changes to the environment (land use and climate change, soil, water and biodiversity depletion). The benefit of this change in perspective is substantial in that it provides the framework for gauging the significant and dynamic role of the livestock sector in driving global environmental change. This in turn should assist and enhance decision-making on necessary action at all levels, from local to global, from private to public, from individual to corporate and from non-governmental to intergovernmental. Action is required: if, as predicted, the production of meat will double from now to 2050, we need to halve impacts per unit of output to achieve a mere status quo in overall impact.

LEAD has been catalysing such action, supported by the Global Environment Facility (GEF) and other donors, in a range of livestock-induced environmental "hotspots", such as in East and Southeast Asia where solutions are designed for the sustainable management of the very large quantities of livestock waste in intensive animal production, such as in Central America where new procedures are introduced for the payment of environmental services in livestock-based land use, and such as in the United Republic of Tanzania where sustainable wildlife–livestock interactions are designed. Such efforts require decisions on, and enforcement of, suitable policy instruments for enabling stakeholder engagement in economically sustainable resource use that addresses the environmental concerns at stake.

It is obvious that the responsibility for the necessary action to address the environmental damage by the livestock sector goes far beyond the sector; it also goes beyond agriculture. While the sector, and agriculture as a whole, have to live up to the challenge of finding suitable technical solutions for more environmentally sustainable resource use in animal agriculture, the decisions concerning their use clearly transcend agriculture; multisector and multiobjective decision-making is required.

It is hoped that this assessment contributes to such decision-making and to thus shrink "*Livestock's long shadow*".

Samuel Jutzi
Director
Animal Production and Health Division
FAO

Contents

Tables

Figures

Boxes

Maps

Acknowledgements

This assessment of global livestock–environment interactions was called for by the Steering Committee of the Livestock, Environment and Development (LEAD) Initiative at its meeting in May 2005 in Copenhagen. The assessment was conducted by the members of the LEAD team at FAO and the chair of LEAD.

This assessment would not have been possible without the financial support and guidance from the LEAD Steering Committee, including Hanne Carus, Jorgen Henriksen and Jorgen Madsen (Denmark), Andreas Gerrits and Fritz Schneider (Switzerland), Philippe Chedanne, Jean-Luc François and Laurent Bonneau (France), Annette von Lossau (Germany), Luis Cardoso (Portugal), Peter Bazeley (United Kingdom), Joyce Turk (United States), Ibrahim Muhammad (Tropical Agricultural Research and Higher Education Center, CATIE), Emmanuel Camus (*Centre de coopération internationale en recherche agronomique pour le développement,* CIRAD), Philippe Steinmetz and Philippe Vialatte (European Union), Samuel Jutzi (Food and Agriculture Organization, FAO), Ahmed Sidahmed (then International Fund for Agriculture, IFAD), Carlos Seré and Shirley Tarawali (International Livestock Research Institute, ILRI), Deborah Bossio (International Water Management Institute, IWMI), Carlos Pomerada (Costa Rica), Modibo Traoré (African Union/Inter-African Bureau for Animal Resources, AU/IBAR), Bingsheng Ke (Research Center for Rural Economy – Ministry of Agriculture, China) and Paul Ndiaye (Université Cheikh Anta-Diop, Senegal).

Our sincere thanks go to those who kindly agreed to review various drafts including Wally Falcon and Hal Mooney (Stanford University, United States), Samuel Jutzi and Freddie Nachtergaele (FAO), Harald Menzi and Fritz Schneider (Swiss College of Agriculture), Andreas Gerrits (Swiss Agency for Development and Cooperation, SDC), Jorgen Henriksen (Denmark) and Günter Fischer (International Institute for Applied Systems Analysis, IIASA), José Martinez (*Institut de recherche pour l'ingéniere de l'agriculture et de l'environnement,* CEMAGREF), Jim Galloway (University of Virginia) and Padma Kumar (Capitalisation of Livestock Programme Experiences in India, CALPI). From within FAO, comments were received from Jelle Bruinsma, Neela Gangadharan, Wulf Killmann and Jan Poulisse. Our thanks also go to Wally Falcon, Hal Mooney and Roz Naylor (Stanford University) for providing a stimulating working environment and continuous debate and encouragement.

We also wish to acknowledge the support of Paul Harrison for style editing; Rosemary Allison for copyediting; Sébastien Pesseat and Claudia Ciarlantini for graphic design; Carolyn Opio, Jan Groenewold and Tom Misselbrook for support in data analysis; Alessandra Falcucci for support in spatial analysis and mapping and Christine Ellefson for a variety of support tasks.

No need to say that all remaining errors and omissions remain the sole responsibility of the authors.

Abbreviations and acronyms

A/R	Afforestation or reforestation
AET	Actual evapotranspiration
ASA	American Soybean Association
AU-IBAR	African Union – Inter-African Bureau for Animal Resources
BMWS	Barley, maize, wheat and soybean
BNF	Biological nitrogen fixation
BOD	Biological oxygen demand
BSE	Bovine spongiform encephalopathy
CALPI	Capitalisation of Livestock Programme Experiences in India
CAP	Common Agricultural Policy
CATIE	Tropical Agricultural Research and Higher Education Centre
CBD	Convention on Biological Diversity
CDM	Clean development mechanism
CEMAGREF	Recherche et expertise sur la multifonctionnalité de l'agriculture
CERs	Certified emissions reductions
CIRAD	Centre de coopération en recherche agronomique pour le développement
CIS	Commonwealth of Independent States
COD	Chemical oxygen demand
CSA	Central and South America
DANIDA	Danish International Development Agency
Embrapa	Empresa Brasileira de Pesquisa Agropecuária – Ministério da Agricoltura, Pecuària e Abastecimento
EU	European Union
FAO	Food and Agriculture Organization of the United Nations
FAOSTAT	FAO statistical databases
FRA	Global Forest Resource Assessment
GATT	General Agreement on Tariffs and Trade
GDP	Gross domestic product
GEF	Global Environmental Facility
GHG	Greenhouse gases
GMO	Genetically modified organisms

GWP	Global warming potential
HPAI	Highly pathogenic avian influenza
IFA	International Fertilizer Industry Association
IFAD	International Fund for Agricultural Development
IFPRI	International Food Policy Research Institute
IIASA	Institute for Applied Systems Analysis
IOM	Institute of Medicine
IPCC	Intergovernmental Panel on Climate Change
IUCN	The World Conservation Union (formerly the International Union for the Conservation of Nature and Natural Resources)
IWMI	International Water Management Institute
LEAD	Livestock, Environment and Development (Initiative)
LPS	Livestock production system
LULUCF	Land use, land-use change and forestry
LWMEAP	Livestock Waste Management in East Asia Project
MAFF–UK	Ministry of Agriculture, Fisheries and Food, United Kingdom of Great Britain and Northern Ireland
MAF–NZ	Ministry of Agriculture and Forestry – New Zealand
MEA	Millennium Ecosystem Assessment
NASA	National Aeronautics and Space Administration
NEC	National Emission Ceiling (directive)
NOAA	National Oceanic and Atmospheric Administration
OECD	Organisation for Economic Co-operation and Development
OIE	World Organization for Animal Health
PES	Payment for environmental services
ppb	Parts per billion
ppm	Parts per million
RCRE	Rutgers Cooperative research and extension
SAfMA	South African Millennium Ecosystem Assessment
SCOPE	Scientific Committee on Problems of the Environment
SOC	Soil organic carbon
SSA	Sub-Saharan Africa
TOC	Total organic carbon
UNCCD	United Nations Convention to Combat Desertification in those Countries Experiencing Serious Drought and/or Desertification, particularly in Africa.
UNCED	United Nations Conference on Environment and Development
UNDP	United Nations Development Programme

UNEP	United Nations Environment Programme
UNEP-WCMC	UNEP World Conservation Monitoring Centre
UNESCO	United Nations Educational, Scientific and Cultural Organization
UNFCCC	United Nations Framework Convention on Climate Change
USDA/FAS	United States Department of Agriculture: Foreign Agricultural Service
USDA-NRCS	United States Department of Agriculture–National Resources Conservation Service
USEPA	United States Environmental Protection Agency
WANA	West Asia and North Africa
WHO	World Health Organization
WMAs	Wildlife Management Areas
WRI	World Resources Institute
WTO	World Trade Organization
WWF	World Wide Fund for Nature

Executive summary

This report aims to assess the full impact of the livestock sector on environmental problems, along with potential technical and policy approaches to mitigation. The assessment is based on the most recent and complete data available, taking into account direct impacts, along with the impacts of feedcrop agriculture required for livestock production.

The livestock sector emerges as one of the top two or three most significant contributors to the most serious environmental problems, at every scale from local to global. The findings of this report suggest that it should be a major policy focus when dealing with problems of land degradation, climate change and air pollution, water shortage and water pollution and loss of biodiversity.

Livestock's contribution to environmental problems is on a massive scale and its potential contribution to their solution is equally large. The impact is so significant that it needs to be addressed with urgency. Major reductions in impact could be achieved at reasonable cost.

Global importance of the sector

Although economically not a major global player, the livestock sector is socially and politically very significant. It accounts for 40 percent of agricultural gross domestic product (GDP). It employs 1.3 billion people and creates livelihoods for one billion of the world's poor. Livestock products provide one-third of humanity's protein intake, and are a contributing cause of obesity and a potential remedy for undernourishment.

Growing populations and incomes, along with changing food preferences, are rapidly increasing demand for livestock products, while globalization is boosting trade in livestock inputs and products. Global production of meat is projected to more than double from 229 million tonnes in 1999/01 to 465 million tonnes in 2050, and that of milk to grow from 580 to 1 043 million tonnes. The environmental impact per unit of livestock production must be cut by half, just to avoid increasing the level of damage beyond its present level.

Structural changes and their impact

The livestock sector is undergoing a complex process of technical and geographical change, which is shifting the balance of environmental problems caused by the sector.

Extensive grazing still occupies and degrades vast areas of land; though there is an increasing trend towards intensification and industrialization. Livestock production is shifting geographically, first from rural areas to urban and peri-urban, to get closer to consumers, then towards the sources of feedstuff, whether these are feedcrop areas, or transport and trade hubs where feed is imported. There is also a shift of species, with production of monogastric species (pigs and poultry, mostly produced in industrial units) growing rapidly, while the growth of ruminant production (cattle, sheep and goats, often raised extensively) slows. Through these shifts, the livestock sector enters into more and direct competition for scarce land, water and other natural resources.

These changes are pushing towards improved efficiency, thus reducing the land area required for livestock production. At the same time, they are marginalizing smallholders and pastoralists, increasing inputs and wastes and increasing and concentrating the pollution created. Widely dispersed non-point sources of pollution are ceding importance to point sources that create more local damage but are more easily regulated.

Land degradation

The livestock sector is by far the single largest anthropogenic user of land. The total area occupied by grazing is equivalent to 26 percent of the ice-free terrestrial surface of the planet. In addition, the total area dedicated to feedcrop production amounts to 33 percent of total arable land. In all, livestock production accounts for 70 percent of all agricultural land and 30 percent of the land surface of the planet.

Expansion of livestock production is a key factor in deforestation, especially in Latin America where the greatest amount of deforestation is occurring – 70 percent of previous forested land in the Amazon is occupied by pastures, and feedcrops cover a large part of the remainder. About 20 percent of the world's pastures and rangelands, with 73 percent of rangelands in dry areas, have been degraded to some extent, mostly through overgrazing, compaction and erosion created by livestock action. The dry lands in particular are affected by these trends, as livestock are often the only source of livelihoods for the people living in these areas.

Overgrazing can be reduced by grazing fees and by removing obstacles to mobility on common property pastures. Land degradation can be limited and reversed through soil conservation methods, silvopastoralism, better management of grazing systems, limits to uncontrolled burning by pastoralists and controlled exclusion from sensitive areas.

Atmosphere and climate

With rising temperatures, rising sea levels, melting icecaps and glaciers, shifting ocean currents and weather patterns, climate change is the most serious challenge facing the human race.

The livestock sector is a major player, responsible for 18 percent of greenhouse gas emissions measured in CO_2 equivalent. This is a higher share than transport.

The livestock sector accounts for 9 percent of anthropogenic CO_2 emissions. The largest share of this derives from land-use changes – especially deforestation – caused by expansion of pastures and arable land for feedcrops. Livestock are responsible for much larger shares of some gases with far higher potential to warm the atmosphere. The sector emits 37 percent of anthropogenic methane (with 23 times the global warming potential (GWP) of CO_2) most of that from enteric fermentation by ruminants. It emits 65 percent of anthropogenic nitrous oxide (with 296 times the GWP of CO_2), the great majority from manure. Livestock are also responsible for almost two-thirds (64 percent) of anthropogenic ammonia emissions, which contribute significantly to acid rain and acidification of ecosystems.

This high level of emissions opens up large opportunities for climate change mitigation through livestock actions. Intensification – in terms of increased productivity both in livestock production and in feedcrop agriculture – can reduce greenhouse gas emissions from deforestation and pasture degradation. In addition, restoring historical losses of soil carbon through conservation tillage, cover crops, agroforestry and other measures could sequester up to 1.3 tonnes of carbon per hectare per year, with additional amounts available through restoration of desertified pastures. Methane emissions can be reduced through improved diets to reduce enteric fermentation, improved manure management and biogas – which also provide renewable energy. Nitrogen emissions can be reduced through improved diets and manure management.

The Kyoto Protocol's clean development mechanism (CDM) can be used to finance the spread of biogas and silvopastoral initiatives involving afforestation and reforestation. Methodologies should be developed so that the CDM can finance other livestock-related options such as soil carbon sequestration through rehabilitation of degraded pastures.

Water

The world is moving towards increasing problems of freshwater shortage, scarcity and depletion, with 64 percent of the world's population expected to live in water-stressed basins by 2025.

The livestock sector is a key player in increasing water use, accounting for over 8 percent of global human water use, mostly for the irrigation of feedcrops. It is probably the largest sectoral source of water pollution, contributing to eutrophication, "dead" zones in coastal areas, degradation of coral reefs, human health problems, emergence of antibiotic resistance and many others. The major sources of pollution are from animal wastes, antibiotics and hormones, chemicals from tanneries, fertilizers and pesticides used for feedcrops, and sediments from eroded pastures. Global figures are not available but in the United States, with the world's fourth largest land area, livestock are responsible for an estimated 55 percent of erosion and sediment, 37 percent of pesticide use, 50 percent of antibiotic use, and a third of the loads of nitrogen and phosphorus into freshwater resources.

Livestock also affect the replenishment of freshwater by compacting soil, reducing infiltration, degrading the banks of watercourses, drying up floodplains and lowering water tables. Livestock's contribution to deforestation also increases runoff and reduces dry season flows.

Water use can be reduced through improving the efficiency of irrigation systems. Livestock's impact on erosion, sedimentation and water regulation can be addressed by measures against land degradation. Pollution can be tackled through better management of animal waste in industrial production units, better diets to improve nutrient absorption, improved manure management (including biogas) and better use of processed manure on croplands. Industrial livestock production should be decentralized to accessible croplands where wastes can be recycled without overloading soils and freshwater.

Policy measures that would help in reducing water use and pollution include full cost pricing of water (to cover supply costs, as well as economic and environmental externalities), regulatory frameworks for limiting inputs and scale, specifying required equipment and discharge levels, zoning regulations and taxes to discourage large-scale concentrations close to cities, as well as the development of secure water rights and water markets, and participatory management of watersheds.

Biodiversity

We are in an era of unprecedented threats to biodiversity. The loss of species is estimated to be running 50 to 500 times higher than background rates found in the fossil record. Fifteen out of 24 important ecosystem services are assessed to be in decline.

Livestock now account for about 20 percent of the total terrestrial animal biomass, and the 30 percent of the earth's land surface that they now pre-empt was once habitat for wildlife. Indeed, the livestock sector may well be the leading player in the reduction of biodiversity, since it is the major driver of deforestation, as well as one of the leading drivers of land degradation, pollution, climate change, overfishing, sedimentation of coastal areas and facilitation of invasions by alien species. In addition, resource conflicts with pastoralists threaten species of wild predators and also protected areas close to pastures. Meanwhile in developed regions, especially Europe, pastures had become a location of diverse long-established types of ecosystem, many of which are now threatened by pasture abandonment.

Some 306 of the 825 terrestrial ecoregions identified by the Worldwide Fund for Nature (WWF) – ranged across all biomes and all biogeographical realms, reported livestock as one of the current threats. Conservation International has identified 35 global hotspots for biodiversity, characterized by exceptional levels of plant endemism and serious levels of habitat loss. Of these, 23 are reported to be affected by livestock production. An analysis of the authoritative World Conservation Union (IUCN) Red List of Threatened Species shows that most of the world's threatened species are suffering habitat loss where livestock are a factor.

Since many of livestock's threats to biodiversity arise from their impact on the main resource sectors (climate, air and water pollution, land degradation and deforestation), major options for mitigation are detailed in those sections. There is also scope for improving pastoralists' interactions with wildlife and parks and raising wildlife species in livestock enterprises.

Reduction of the wildlife area pre-empted by livestock can be achieved by intensification. Protection of wild areas, buffer zones, conservation easements, tax credits and penalties can increase the amount of land where biodiversity conservation is prioritized. Efforts should extend more widely to integrate livestock production and producers into landscape management.

Cross-cutting policy frameworks

Certain general policy approaches cut across all the above fields. A general conclusion is that improving the resource use efficiency of livestock production can reduce environmental impacts.

While regulating about scale, inputs, wastes and so on can help, a crucial element in achieving greater efficiency is the correct pricing of natural resources such as land, water and use of waste sinks. Most frequently natural resources are free or underpriced, which leads to overexploitation and pollution. Often perverse subsidies directly encourage livestock producers to engage in environmentally damaging activities.

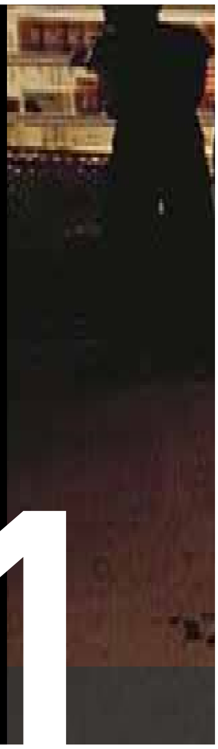
A top priority is to achieve prices and fees that reflect the full economic and environmental costs, including all externalities. One requirement for prices to influence behaviour is that there should be secure and if possible tradable rights to water, land, use of common land and waste sinks.

Damaging subsidies should be removed, and economic and environmental externalities should be built into prices by selective taxing of and/or fees for resource use, inputs and wastes. In some cases direct incentives may be needed.

Payment for environmental services is an important framework, especially in relation to extensive grazing systems: herders, producers and landowners can be paid for specific environmental services such as regulation of water flows, soil conservation, conservation of natural landscape and wildlife habitats, or carbon sequestration. Provision of environmental services may emerge as a major purpose of extensive grassland-based production systems.

An important general lesson is that the livestock sector has such deep and wide-ranging environmental impacts that it should rank as one of the leading focuses for environmental policy: efforts here can produce large and multiple payoffs. Indeed, as societies develop, it is likely that environmental considerations, along with human health issues, will become the dominant policy considerations for the sector.

Finally, there is an urgent need to develop suitable institutional and policy frameworks, at local, national and international levels, for the suggested changes to occur. This will require strong political commitment, and increased knowledge and awareness of the environmental risks of continuing "business as usual" and the environmental benefits of actions in the livestock sector.



01

Introduction

Livestock activities have significant impact on virtually all aspects of the environment, including air and climate change, land and soil, water and biodiversity. The impact may be direct, through grazing for example, or indirect, such as the expansion of soybean production for feed replacing forests in South America.

Livestock's impact on the environment is already huge, and it is growing and rapidly changing. Global demand for meat, milk and eggs is fast increasing, driven by rising incomes, growing populations and urbanization.

As an economic activity, livestock production is technically extremely diverse. In countries or areas where there is no strong demand for food products of animal origin, subsistence and low-input production prevails, mainly for subsistence rather than for commercial purposes. This contrasts with commercial, high-input production in areas serving a growing or established high demand. Such diverse production systems make extremely diverse claims on resources. The diversity of production systems and interactions makes the analysis of the livestock–environment interface complex and sometimes controversial.

The livestock sector affects a vast range of natural resources, and must be carefully managed given the increasing scarcity of these resources and the opportunities that they represent for other sectors and activities. While intensive livestock production is booming in large emerging countries, there are still vast areas where extensive livestock production and its associated livelihoods persist. Both intensive and extensive production requires attention and intervention so that the livestock sector can have fewer negative and more positive impacts on national and global public goods.

A major motivation for this assessment is that the environmental issues linked to livestock have not generally received an adequate institutional response – neither in developing nor in developed countries. Livestock sector growth in some places, and stagnation with poverty in others, go largely uncontrolled. Although usually considered part of agriculture, in many places livestock production has grown in the same way as industry, and is no longer directly tied to land or to specific locations.

As the environment around the animals is increasingly modified and standardized, environmental impacts swiftly change. Public policies, in developed and developing countries alike, barely keep pace with rapid transformations in production technology and structural shifts in the sector. Environmental laws and programmes are usually put in place only after significant damage has already occurred. The focus continues to be placed on protection and restoration, rather than on the more cost-effective approaches of prevention and mitigation.

In the varied contexts of the livestock sector, environmental issues require an integrated approach, combining policy measures and technology changes, within a framework of multiple objectives.

The livelihood concerns of hundreds of millions of poor livestock holders, who often engage in livestock production because they have no alternative, must be taken into account. The demands of the emerging middle class, who are consuming growing amounts of meat, milk and eggs, cannot be ignored either. Attempts to curb the booming demand for these products have generally proved ineffective.

Better policies in the livestock sector are an environmental requirement, and a social and health necessity. Animal foods are susceptible to pathogens and often carry chemical residues. Food safety requirements must be met, and are generally a prerequisite in formal markets.

The previous assessments of the Livestock Environment and Development (LEAD) Initiative (de Haan, Steinfeld and Blackburn, 1997; Steinfeld, de Haan and Blackburn, 1997) emphasized the livestock sector perspective and analysed livestock-environment interactions from the perspective of a livestock production system.

This updated assessment inverts this approach and starts from an environmental perspective. It attempts to provide an objective assessment of the many diverse livestock–environment interactions. Economic, social and public health objectives are of course taken into account so as to reach realistic conclusions. This assessment then outlines a series of potential solutions that can effectively address the negative consequences of livestock production.

1.1 Livestock as a major player in global environmental issues

Livestock have a substantial impact on the world's water, land and biodiversity resources and contribute significantly to climate change.

Directly and indirectly, through grazing and through feedcrop production, the livestock sector occupies about 30 percent of the ice-free terrestrial surface on the planet. In many situations, livestock are a major source of land-based pollution, emitting nutrients and organic matter, pathogens and drug residues into rivers, lakes and coastal seas. Animals and their wastes emit gases, some of which contribute to climate change, as do land-use changes caused by demand for feedgrains and grazing land. Livestock shape entire landscapes and their demands on land for pasture and feedcrop production modify and reduce natural habitats.

Using animals for food and other products and services is only one of many human activities that depend on natural resources. Humans are using the world's renewable natural resources at rates that increasingly exceed their natural abilities to renew themselves (Westing, Fox and Renner, 2001). Humans introduce growing amounts of pollutants into the air, water and soil, at rates ever higher than the capacity of the environment to dissipate or decompose these pollutants. Humans are encroaching on what remains of relatively undisturbed environments, putting biodiversity at risk of mass extinction. Anthropogenic land-use changes have accelerated over the last decades, most dramatically in developing countries. Urbanization and expansion of cropping have led to an unprecedented loss and fragmentation of habitats, including valuable ones such as forests and wetlands.

Water availability is becoming a serious constraint to the expansion of agriculture and to meeting other growing human needs. Agriculture is the largest user of water, accounting for 70 percent of total freshwater use.

While there are different views on the extent of climate change and its effect on the environment, it is now firmly established that anthropogenic climate change is indeed occurring. The most important gas associated with climate change is carbon dioxide (CO_2) while other greenhouse gases, including methane, nitrous oxide, ozone and sulphur hexafluoride also contribute. Carbon dioxide levels have increased by over 40 percent over the past 200 years, from 270 parts per million (ppm) to 382 ppm (NOAA, 2006). Today, CO_2 concentrations are higher than at any time during the last 650 000 years (Siegenthaler *et al*., 2005). Methane concentrations today are more than twice the pre-industrial level (Spahni *et al*., 2005). Average temperatures have increased by 0.8°C over the past century (NASA, 2005). Combustion of fossil fuels is a major contributor to these changes.

Climate change means an increase in average temperature and seems to be associated with an increased frequency of extreme weather events. FAO warns that food distribution systems and their infrastructure will be disrupted, and this may greatly increase the number of hungry people, most severely in sub-Saharan Africa (FAO, 2005a). According to FAO, developing countries may lose about 280 million tonnes of potential cereal production as a result of climate change.

Because of habitat losses, unsustainable forms of exploitation and climate change, the loss of biodiversity continues to accelerate. The Millennium Ecosystem Assessment (MEA, 2005a), in a comprehensive assessment of the environmental health of the planet, estimates that species are disappearing at 100 to 1 000 times the background levels seen in fossil records. The MEA gauged that one-third of all amphibians, a fifth of mammals and an eighth of all birds are now threatened by extinction. This assessment is based on known species and it is estimated that 90 percent or more of all existing species have not been catalogued yet. While some species provide obvious services such as food, timber or clothing, most species' services are more difficult to see and, therefore, less appreciated. They include recycling of nutrients, pollination and seed dispersal, climate control and purification of air and water.

Additional land available for cultivation is limited. Therefore, most of the increase in agricultural production has come, and will come, from intensification of land that is already cropped or grazed. As a large user of crops and other plant material, the livestock sector must continue to improve the conversion of these materials into edible products.

The overall impact of livestock activities on the environment is enormous. Part of the damage can be offset by applying scientific knowledge and technological capability for dealing with these problems. Meanwhile, the vast legacy of damage leaves future generations with a debt. Ultimately, environmental issues are social issues: environmental costs created by some groups and nations are carried by others, or by the planet as a whole. The health of the environment and the availability of resources affect the welfare of future generations, and overuse of resources and excess environmental pollution by current generations are to their detriment.

Environmental degradation is often associated with war and other forms of conflict. Throughout history, peoples and nations have fought over natural resources such as land and water. By increasing the scarcity of these resources, environmental degradation increases the likelihood of violent conflict, particularly when there is a lack of governing institutions. In recent years, public attention has been drawn to the prospect that future wars will be fought over increasingly scarce natural resources (see, for example, Klare, 2001, or Renner, 2002). A Pentagon report (Schwartz and Randall, 2003) suggested that global warming could prove a greater risk to the world than terrorism and could lead to catastrophic droughts, famines and riots.

At the local or regional level, the Southern African Millennium Ecosystem Assessment (SAfMA) (Biggs *et al.*, 2004) reveals a striking connection between ecological stress and social conflict. This SAfMA study suggests causal links in both directions; conflict may cause environmental degradation but the latter may also trigger conflict. The study quotes political violence in South Africa's KwaZulu Natal Province as an example where faction fighting over scarce land for cattle has led to a series of killings. Water scarcity, land degradation from overgrazing or woodfuel shortages can also lead to conflict. The same study points to Burundi, Rwanda and eastern Congo as areas where major ecological problems have marched hand in hand with recent histories of violent conflict.

Environmental degradation significantly affects human health, both directly and indirectly. Direct effects on human health include contact with pollutants. Indirect effects include increased exposure of humans and of animals to infectious diseases because of climate change. The geographic range and seasonality of a number of important diseases, including malaria and dengue fever, are very sensitive to changes in climatic conditions (UNEP 2005a). Schistosomiasis or bilharzia, carried by water snails, is associated with changing water flows. The World Resources Report (1999) underlines how the burden of these preventable and environment-related diseases is borne disproportionately by the poor, both in developing and developed countries.

Environmental degradation at its current scale and pace is clearly a serious threat for the sustainability of natural resources. The functioning of ecosystems, both at local and global levels, is already seriously compromised. Ultimately, if left unchecked, environmental degradation may threaten not only economic growth and stability but the very survival of humans on the planet.

1.2 The setting: factors shaping the livestock sector

The livestock sector, along with food and agriculture in general, is undergoing far-reaching change, much of it driven by factors outside the sector. Growing populations and other demographic factors such as age structure and urbanization determine food demand and have driven the intensification of agriculture for centuries. Growing economies and individual incomes have also contributed to growing demand and a shift in diets. These trends have accelerated over the last two decades in large parts of Asia, Latin America and the Near East, spurring a rapid increase in demand for animal products and other high value foodstuffs such as fish, vegetables and oils.

The agriculture sector has responded to the increased and diversified demands for food items with innovations in biology, chemistry and machinery. It has done so mainly through intensification rather than expansion. Land use has changed correspondingly.

These secular changes of population, economies, diets, technology and land use drive changes in the global livestock sector while, to some extent, the sector itself shapes these forces. Sketching these broad developments helps to understand the context within which the livestock sector operates.

The demographic transition

Growing populations and cities boost and change food demand

Population and population growth are major determinants of the demand for food and other agricultural products. World population is currently 6.5 billion, growing at the rate of 76 million annually (UN, 2005). The UN's medium projection forecasts that world population will reach 9.1 billion by 2050, peaking at around 9.5 billion by the year 2070 (UN, 2005).

While populations in the developed countries as a whole are close to stagnant, 95 percent of the population increase is occurring in developing countries. The fastest population growth rates (averaging 2.4 percent annually) are occurring in the group of 50 least developed countries (UN, 2005). Population growth rates are slowing because of decreased fertility rates, and are below replacement levels in most developed countries and decreasing rapidly in emerging countries, although they remain high in least developed countries.

Fertility decline, in conjunction with increases in life expectancy, is leading to population ageing globally. The proportion of older people (aged 60 and over) is projected to double to more than 20 percent from today's level (UN, 2005). Age groups differ in their dietary and consumption patterns, with adults and older people typically consuming larger amounts of animal protein than children.

Another important factor determining demand for food is urbanization. In 2005 (the latest year for which statistics are available) 49 percent of the world population were living in cities (FAO, 2006b). This global figure masks important differences among the world regions: sub-Saharan Africa and South Asia are still only moderately urbanized – with 37 and 29 percent urbanization, respectively – whereas urbanization rates are around 70 to 80 percent in developed countries and in Latin America (FAO, 2006a; 2006b) (see Table 1.1).

Urbanization continues in all regions of the world, with growth rates highest where current urbanization is low, particularly in South Asia and sub-Saharan Africa. Virtually all population growth between 2000 and 2030 will be urban (FAO, 2003a) (see Figure 1.1).

Urbanization usually implies higher levels of participation in the workforce and has an impact on patterns of food consumption, In cities, people typically consume more food away from home, and consume higher amounts of pre-cooked, fast and convenience foods, and snacks

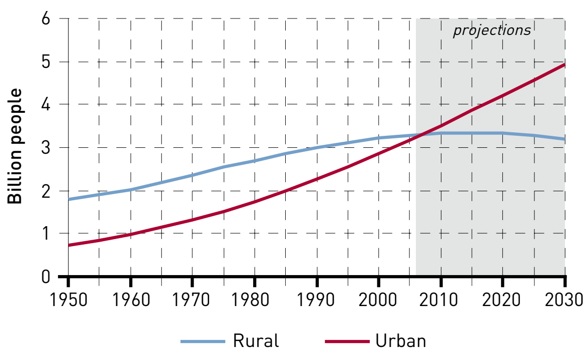
Table 1.1

Urbanization rates and urbanization growth rates

Region	Urban population as percent of total population in 2005	Urbanization growth rate *(Percentage per annum 1991–2005)*
South Asia	29	2.8
East Asia and the Pacific	57	2.4
Sub-Saharan Africa	37	4.4
West Asia and North Africa	59	2.8
Latin America and the Caribbean	78	2.1
Developing countries	57	3.1
Developed countries	73	0.6
World	49	2.2

Source: FAO (2006a) and FAO (2006b).

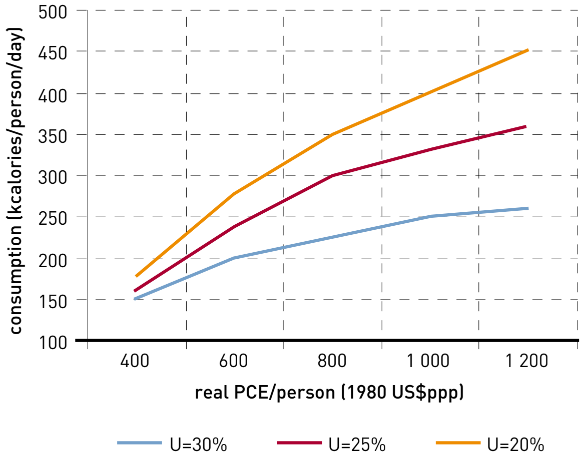
Figure 1.1 **Past and projected global rural and urban populations from 1950 to 2030**



Source: FAO (2006a) and FAO (2006b).

A student orders fast food near Luve – Swaziland

Figure 1.2 **Consumption function of animal products at different levels of urbanization in China**

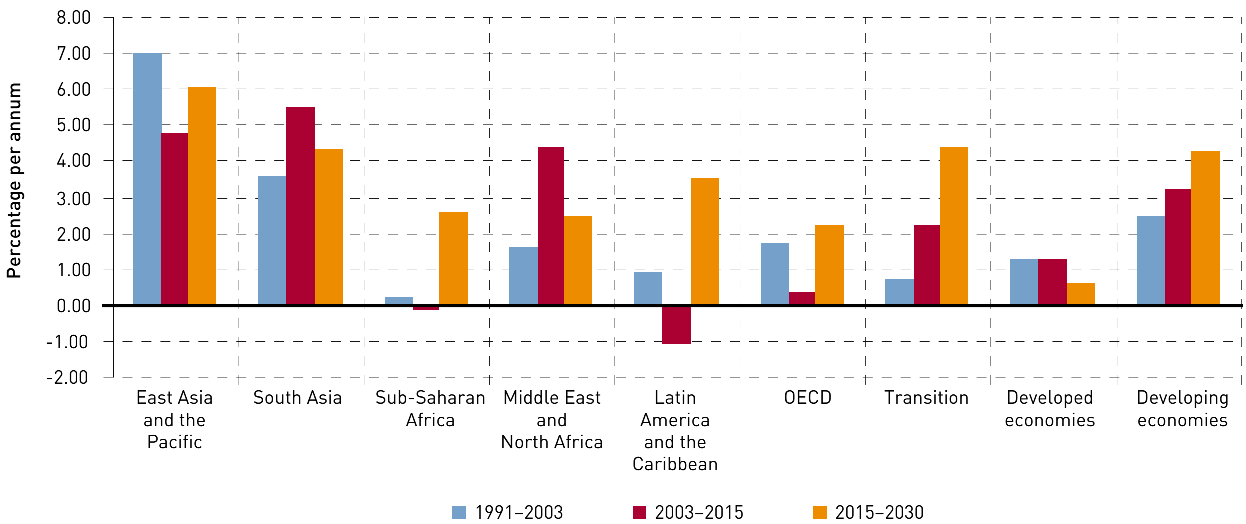


Note: PCE: per capita expenditure. U = percent urban. PPP: purchasing power parity.
Source: Rae (1998).

(Schmidhuber and Shetty, 2005; Rae, 1998; King, Tietyen and Vickner, 2000). Therefore, urbanization influences the position and the shape of the consumption functions for animal products (Rae, 1998) – this function measures the way in which consumption of a given item responds to changes in total expenditure.

For China, a given increase in urbanization has a positive effect on per capita consumption levels of animal products (Rae, 1998) (Figure 1.2). Between 1981 and 2001, human consumption of grains dropped by 7 percent in rural areas of China and 45 percent in urban areas. Meanwhile, meat and egg consumption increased by 85 percent and 278 percent respectively in rural areas and by 29 percent and 113 percent in urban areas (Zhou, Wu and Tian, 2003).

Figure 1.3 **Past and projected GDP per capita growth by region**



Source: World Bank (2006) and FAO (2006a).

Economic growth

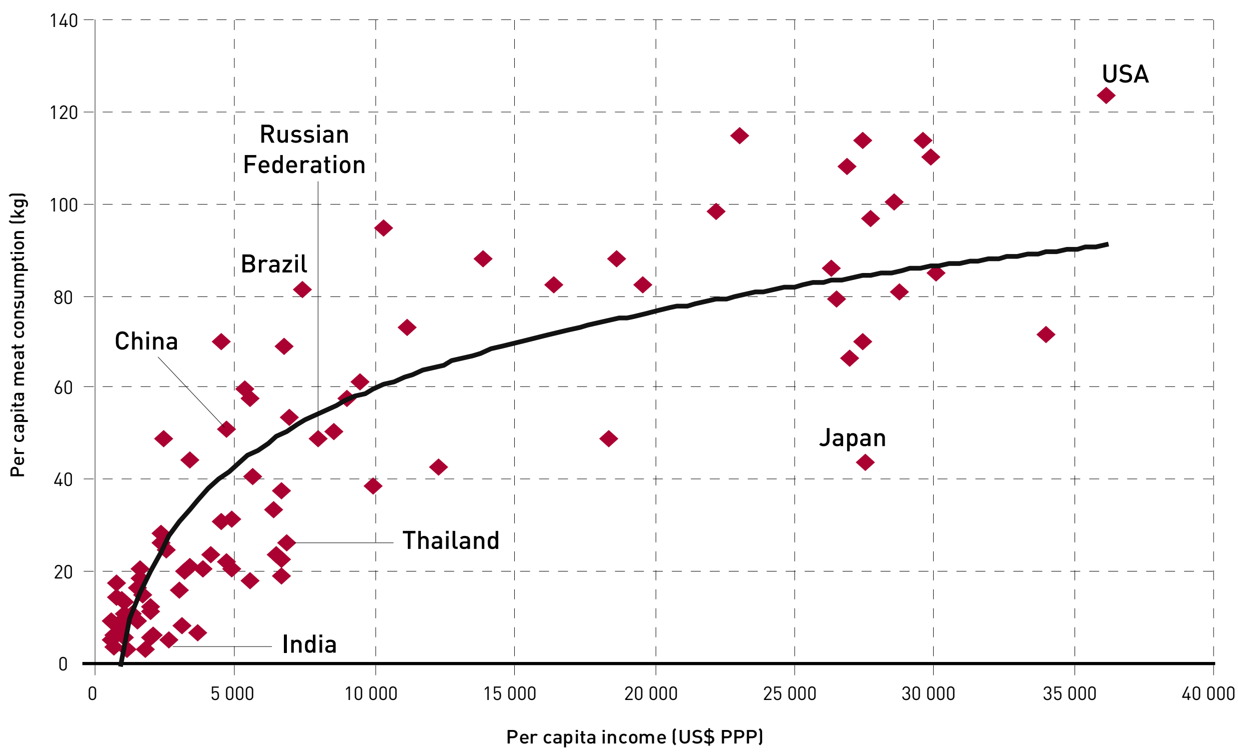
Growing incomes boost demand for livestock products

Over recent decades, the global economy has experienced an unparalleled expansion. Population growth, technological and science breakthroughs, political changes, and economic and trade liberalization have all contributed to economic growth. In developing countries, this growth has translated into rising per capita incomes, and an emerging middle class that has purchasing power beyond their basic needs.

Over the decade 1991 to 2001, per capita GDP grew at more than 1.4 percent a year for the world as a whole. Developing countries grew at 2.3 percent on average compared to 1.8 percent for developed countries (World Bank, 2006). Growth has been particularly pronounced in East Asia with an annual growth rate of close to seven percent, led by China, followed by South Asia with 3.6 percent. The World Bank (2006) projects that GDP growth in developing countries will accelerate in coming decades (Figure 1.3).

There is a high income elasticity of demand for meat and other livestock products (Delgado *et al.*, 1999) – that is, as incomes grow, expenditure on livestock products grows rapidly. Therefore growing per capita incomes will translate into growing demand for these products. This will close much of the gap in average consumption figures of meat, milk and eggs that currently exists between developed and developing countries. As Figure 1.4 shows, the effect of increased income on diets is greatest among lower- and middle-income populations. This observation is true at individual level as well as at the national level (Devine, 2003).

Figure 1.4 The relationship between meat consumption and per capita income in 2002



Note: National per capita based on purchasing power parity (PPP).
Source: World Bank (2006) and FAO (2006b).

The nutrition transition

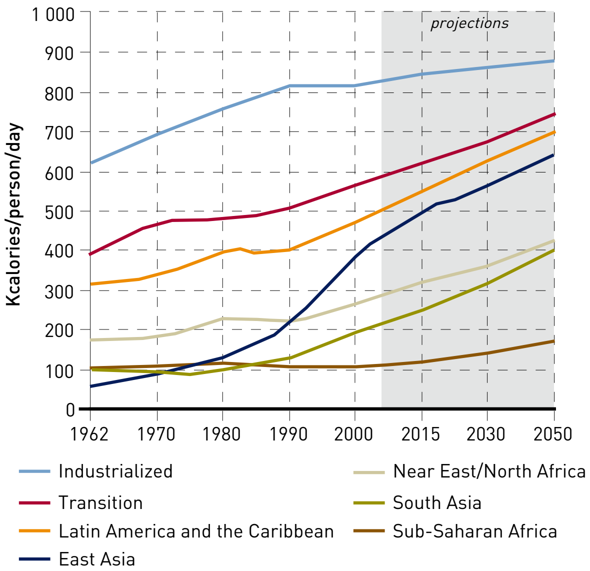
Worldwide shifts in dietary preferences

The advent of agriculture and the sedentarization of hunter/gatherers enabled increasing populations to be fed – but it also led to a narrowing of the human diet. Prior to agriculture, animal products played a much larger role in human nutrition, and intake levels were similar to, if not higher than, current consumption levels in developed countries. Increases in income and advances in agriculture enabled developed countries to enrich and diversify their diets over the last 150 years. Developing countries are currently engaged in a catching-up process, which has been termed the "nutrition transition" by Popkins, Horton and Kim, (2001). The transition is characterized by an accelerated shift from widespread undernourishment to richer and more varied diets and often to overnutrition. In contrast to the more secular nutrition transition that occurred in developed countries, this shift now occurs within a single generation in rapidly growing developing countries.

With higher disposable incomes and urbanization, people move away from relatively monotonous diets of varying nutritional quality (based on indigenous staple grains or starchy roots, locally grown vegetables, other vegetables and fruits, and limited foods of animal origin) towards more varied diets that include more pre-processed food, more foods of animal origin, more added sugar and fat, and often more alcohol (Table 1.2 and Figure 1.5). This shift is accompanied by reduced physical activity, leading to a rapid increase in overweight and obesity (Popkin, Horton and Kim, 2001). Worldwide the number of overweight people (about 1 billion) has now surpassed the number of malnourished people (about 800 million). And a significant part of the growth in obesity occurs in the developing world. For example, the World Health Organization (WHO) estimates that there are 300 million obese adults and 115 million suffering from obesity-related conditions in the developing world.[1] A rapid increase in diet-related chronic diseases, including heart disease, diabetes, hypertension and certain cancers is associated with the rapid nutrition transition. In a number of developing countries, diet-related chronic diseases have become a priority in national food and agricultural policies, which now promote healthy eating habits, exercise and school-based nutrition programmes (Popkin, Horton and Kim, 2001).

The nutrition transition is driven by rising

Figure 1.5 **Past and projected food consumption of livestock products**



Note: For past, three-year averages centered on the indicated year. Livestock products include meats, eggs, milk and dairy products (excluding butter).
Source: FAO (2006a) and FAO (2006b).

Table 1.2

Changes in food consumption in developing countries

	1962	1970	1980	1990	2000	2003
	Consumption kg/person/year					
Cereals	132	145	159	170	161	156
Roots and tubers	18	19	17	14	15	15
Starchy roots	70	73	63	53	61	61
Meat	10	11	14	19	27	29
Milk	28	29	34	38	45	48

Source: FAO (2006b).

[1] Available at: www.fao.org/FOCUS/E/obesity/obes1.htm

incomes and by the continuing trend to lower relative prices for food. Prices have been declining in real terms since the 1950s. Currently they allow much higher consumption levels of high-value food items than was the case for developed countries at comparable levels of income in the past (Schmidhuber and Shetty, 2005).

While purchasing power and urbanization explain the greater part of the per capita consumption pattern, other social and cultural factors can have a large influence locally. For example, Brazil and Thailand have similar income per capita and urbanization rates but animal product consumption in Brazil is roughly twice as high as in Thailand. The Russian Federation and Japan have similar consumption levels for animal-derived foods, yet income levels in Japan are about 13 times higher than in Russia (see figure 1.4).

Natural resource endowment is one of the additional factors determining consumption, as it shapes the relative costs of different food commodities. Access to marine resources, on the one hand, and to natural resources for livestock production, on the other, have drawn consumption trends in opposite directions. Lactose intolerance, found particularly in East Asia, has limited milk consumption. Cultural reasons have further influenced consumption habits. This, for example, is the case in South Asia, where consumption per capita of meat is lower than income alone would explain. Other examples are the exclusion of pork from the diet by Muslims. Socio-cultural patterns have created a rich diversity of consumer preferences, but have also influenced consumers' views about the quality of animal products (Krystallis and Arvanitoyannis, 2006).

More recently, consumption patterns are increasingly influenced by growing concerns about health, the environment, ethical, animal welfare and development issues. In countries of the Organisation for Economic Co-operation and Development (OECD) a class of "concerned consumers" has emerged, who (Harrington, 1994) tend to reduce their consumption of livestock food products or opt for certified products, such as free range or organic foods (Krystallis and Arvanitoyannis, 2006; King, *et al.*, 2000). The growing trend towards vegetarianism, albeit still at a very low level in most societies, is another manifestation of this trend. Government promotion campaigns are also identified as potential drivers of consumption trends (Morrison *et al.*, 2003).

Technological change

Growing productivity

The livestock sector has been affected by profound technological change on three different fronts:

- In livestock production, the widespread application of advanced breeding and feeding technology has spurred impressive productivity growth in most parts of the world.
- In crop agriculture, irrigation and fertilization techniques, combined with the use of improved varieties and mechanization, continue to translate into growing yields and improved nutrient composition in pasture and major crops used for feed.
- The application of modern information technology and other technical changes are improving post-harvest, distribution and marketing of animal products.

In animal production, technological development has been most rapid in those subsectors that have experienced the fastest growth: broiler and egg production, pork and dairy. Productivity growth, and the underlying spread of advanced technologies, has been less pronounced for beef and meat from small ruminants. However, certain key technological changes have occurred in the production of all livestock commodities – a growing production intensity, characterized by increasing use of feed cereals, use of advanced genetics and feeding systems, animal health protection and enclosure of animals. Advances in these areas go hand in hand, and it is difficult to separate out the effect of individual factors on overall productivity increases.

Increased grain feeding

Traditionally, livestock production was based on locally available feed resources such as crop wastes and browse that had no value as food. However, as livestock production grows and intensifies, it depends less and less on locally available feed resources, and increasingly on feed concentrates that are traded domestically and internationally. In 2002, a total of 670 million tonnes of cereals were fed to livestock, representing roughly one-third of the global cereal harvest (see Table 1.3). Another 350 million tonnes of protein-rich processing by-products are used as feed (mainly brans, oilcakes and fishmeal).

Monogastric species that can most efficiently make use of concentrate feeds, i.e. pigs, poultry and dairy cattle, have an advantage over beef cattle, sheep and goats. Among the monogastrics, poultry has shown the highest growth rates and lowest prices, mainly because of favourable feed conversion rates. The use of feed concentrate for ruminants is limited to countries where meat prices are high relative to grain prices. Where grain prices are high relative to meat prices – typically in food-deficit developing countries – grain feeding to ruminants is not profitable.

What is driving the increasing use of feed grains? Most importantly, there is a long-term decline of grain prices; a trend that has persisted since the 1950s. Supply has kept up with growing demand: total supply of grains increased by 43 percent over the last 24 years (1980 to 2004). In real terms (constant US$), international prices for grains have halved since 1961. Expanding supply at declining prices has been achieved by area expansion and by intensification of crop production.

Intensification accounts for the bulk of supply expansion over the past 25 years, and is a result of technological advances and higher input use in crop production – notably plant breeding, the application of fertilizers and mechanization. Area expansion has been an important contributor to growing supplies in many developing countries, especially in Latin America (where the cropped area expanded by 15 percent between 1980 and 2003) and sub-Saharan Africa (22 percent). Land-scarce Asia (developing) has seen a modest 12 percent expansion of the cropped area. Some countries have seen a particularly strong expansion of area cropped, most of it at the expense of forest (Brazil and other Latin American countries). Much of this area expansion has been for the production of concentrate feeds for livestock, notably soybeans and maize. Feed conversion and growth rates have been greatly improved by use of linear programming to develop least-cost feed rations, phased feeding and the use of enzymes and synthetic amino-acids, together with a much extended use of feed concentrates (grains and oilcakes).

In future, feed concentrate use is projected to grow more slowly than livestock production, despite the fact that the latter is becoming increasingly cereal-based. This is because improved technologies in feeding, breeding and animal health are producing even greater efficiency gains.

Table 1.3

Use of feed concentrate

	Feed concentrate use in 2002 *(million tonnes)*		
Commodity group	**Developing countries**	**Developed countries**	**World**
Grains	226.4	444.0	670.4
Brans	92.3	37.0	129.3
Oilseeds and pulses	11.6	15.7	27.3
Oilcakes	90.5	96.6	187.3
Roots and tubers	57.8	94.6	152.4
Fish meal	3.8	3.8	7.6
Total of above	**482.4**	**691.71**	**1 174.1**

Source: FAO (2005).

More productive breeds

In animal genetics and breeding, the use of hybridization and artificial insemination has sped up the process of genetic improvement. In poultry, for example, these techniques have greatly expanded the number of animals that can be bred from a superior parent stock, creating animals with uniform characteristics (Fuglie *et al*., 2000). Traditionally, the only means of genetic improvement was selection based on the phenotype. Starting from the beginning of the twentieth century, technologies such as controlled management of reproduction and of pedigrees were developed. These were initially limited to purebred stock (Arthur and Albers, 2003). Around mid-century, line specialization and cross-breeding were initiated, first in North America, then in Europe and other OECD countries. Artificial insemination was first introduced in the 1960s and is now commonplace in all intensive livestock production systems. Around the same time, breeding value evaluation technologies were introduced in developed countries. More recent innovations include the use of DNA markers to identify specific traits.

Breeding goals have changed considerably over time, but the speed and precision with which these goals can be achieved has increased considerably over recent decades. Short-cycle species, such as poultry and pigs, have a distinct advantage over species having a longer generation interval. Among all species, feed conversion and related parameters such as growth rate, milk yield and reproductive efficiency are paramount factors for breeding (Arthur and Albers, 2003). Fat content and other features that correspond best to consumer demands are increasing in importance.

These changes have brought about impressive results. For example, Arthur and Albers (2003) report that in the United States, feed conversion ratios for eggs have been reduced from 2.96 grams of feed per gram of egg in 1960 to 2.01 grams in 2001.

The breeding industry has been less successful in developing breeds of dairy cows, pigs and poultry that perform well in non-modified tropical low-input environments. Highly intensive livestock enterprises in the tropics usually control the climatic and health environment around the animals, so as to utilize the efficiencies of modern breeds developed for temperate regions.

Animal health improvements have further contributed to raising productivity, including the use of antibiotics (now banned for use in growth enhancement in areas such as the European Union (EU) in special pathogen-free production environments. In developing countries, these technologies have spread widely in recent years, particularly in industrial production systems close to major consumption centres. The continuous increase in scales of production has also led to important productivity gains in developing countries. These have allowed animal products to be supplied to growing populations at decreasing real prices (Delgado *et al*., 2006).

Cheaper feed grains

In crop production, similar improvements have improved supply and reduced prices of feed grains, with important productivity increases occurring earlier (in the 1960s and 1970s) than for livestock (FAO, 2003a). For developing countries, about 80 percent of the projected growth in crop production to 2030 will come from intensification, mostly in the form of yield increases, and also through higher cropping intensities. Irrigation is a major factor in land intensification: the irrigated area in developing countries doubled between 1961–63 and 1997–99 and is expected to increase by another 20 percent by 2030 (FAO, 2003a). Widespread application of fertilizer and improved fertilizer composition and forms of application are other important factors in crop intensification, along with improvements in plant protection.

The post-harvest sector, distribution and marketing have seen profound structural changes. These are associated with the emergence of large retailers, with a tendency towards vertical integration and coordination along the food chain. This trend has been brought about by liberaliza-

Table 1.4

Key productivity parameters for livestock in different world regions

Region	Chicken meat (kg output/kg biomass/year)[1]		Egg yield (kg/layer/year)		Pig meat (kg output/kg biomass/year)[1]	
	1980	2005	1980	2005	1980	2005
World	1.83	2.47	8.9	10.3	0.31	0.45
Developing countries	1.29	1.98	5.5	8.8	0.14	0.33
Developed countries	2.26	3.55	12.2	15.0	0.82	1.20
Sub-Saharan Africa	1.46	1.63	3.4	3.6	0.53	0.57
West Asia and North Africa	1.73	2.02	7.0	9.4	1.04	1.03
Latin America and the Caribbean	1.67	3.41	8.6	9.8	0.41	0.79
South Asia	0.61	2.69	5.8	8.1	0.72	0.71
East and Southeast Asia	1.03	1.41	4.7	9.5	0.12	0.31
Industrialized countries	2.45	3.72	14.1	16.0	1.03	1.34
Transition countries	1.81	2.75	9.6	13.0	0.57	0.75

Region	Beef (kg output/kg biomass/year)[1]		Small ruminants (kg output/kg biomass/year)[1]		Milk yield (kg/cow/year)	
	1980	2005	1980	2005	1980	2005
World	0.11	0.13	0.16	0.26	1 974	2 192
Developing countries	0.06	0.09	0.14	0.26	708	1 015
Developed countries	0.17	0.21	0.19	0.24	3 165	4 657
Sub-Saharan Africa	0.06	0.06	0.15	0.15	411	397
West Asia and North Africa	0.07	0.10	0.21	0.25	998	1 735
Latin America and the Caribbean	0.08	0.11	0.11	0.13	1 021	1 380
South Asia	0.03	0.04	0.16	0.23	517	904
East and Southeast Asia including China	0.06	0.16	0.05	0.20	1 193	1 966
Industrialized countries	0.17	0.20	0.20	0.25	4 226	6 350
Transition countries	0.18	0.22	0.17	0.23	2 195	2 754

[1] Biomass is calculated as inventory x average liveweight. Output is given as carcass weight.
Source: FAO (2006b).

tion of markets and widespread application of new technologies in logistics and organizational management transport. All of these help to bring prices down for consumers – but at the same time they raise entry barriers for small producers (Costales, Gerber and Steinfeld, 2006).

1.3 Trends within the livestock sector

Until about the early 1980s, diets that included daily consumption of milk and meat were largely the privilege of OECD country citizens and a small wealthy class elsewhere. At that time, most developing countries, with the exception of Latin America and some West Asian countries had an annual per capita meat consumption of substantially less than 20 kg. For most people in Africa and Asia, meat, milk and eggs were an unaffordable luxury, consumed only on rare occasions. A high proportion of the larger livestock in developing countries was not primarily kept for food, but for other important functions, such as providing draught power and manure, and serving as an insurance policy and a capital asset, usually disposed of only in times of communal feasting or emergency.

This is changing rapidly. The livestock sector is currently growing faster than the rest of agriculture in almost all countries. Typically, its share in agricultural GDP rises with income and level of development and is above 50 percent for most OECD countries. The nature of livestock production is also changing rapidly in many emerging economies, as well as in developed countries. Most of this change can be summarized under the term "industrialization". Through industrialization, livestock escape most of the environmental constraints that have shaped livestock production diversely in the wide range of environments in which they occur.

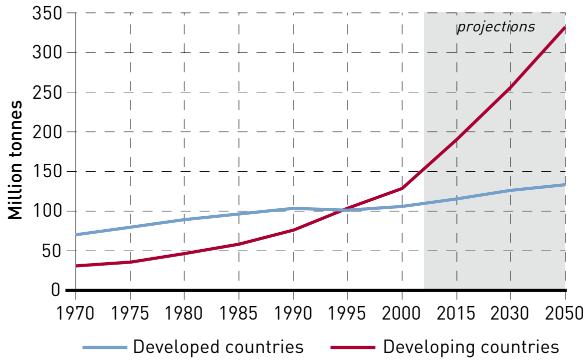
Livestock production and consumption booms in the south, stagnates in the north

Driven by population growth and rising income in many developing countries, the global livestock sector has seen a dramatic expansion over the past decades, though with considerable differences between developing and developed countries.

In the developing countries, annual per capita consumption of meat has doubled since 1980, from 14 kg to 28 kg in 2002 (Table 1.5).

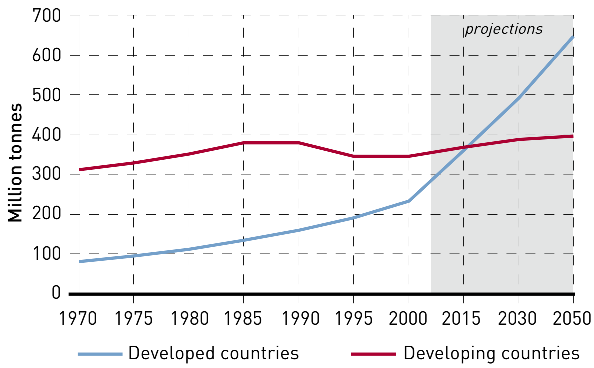
Total meat supply tripled from 47 million tonnes to 137 million tonnes over the same period. Developments have been most dynamic in

Figure 1.6 **Past and projected meat production in developed and developing countries from 1970 to 2050**



Source: FAO (2006a) and FAO (2006b).

Figure 1.7 **Past and projected milk production in developed and developing countries from 1970 to 2050**



Source: FAO (2006a) and FAO (2006b).

Table 1.5

Past and projected trends in consumption of meat and milk in developing and developed countries

	Developing countries					Developed countries				
	1980	1990	2002	2015	2030	1980	1990	2002	2015	2030
Food demand										
Annual per capita meat consumption (kg)	14	18	28	32	37	73	80	78	83	89
Annual per capita milk consumption (kg)	34	38	46	55	66	195	200	202	203	209
Total meat consumption (million tonnes)	47	73	137	184	252	86	100	102	112	121
Total milk consumption (million tonnes)	114	152	222	323	452	228	251	265	273	284

Source: FAO (2006a) and FAO (2006b).

Table 1.6

Developing country trends in livestock production in 2005

Country Group/Country	Meat (million tonnes)	Milk (million tonnes)	Percentage of developing country production	
			Meat	Milk
Developing countries	155.0	274.1	100.0	100.0
China	75.7	28.3	48.8	10.3
Brazil	19.9	23.5	12.8	8.6
India	6.3	91.9	4.1	33.5

Source: FAO (2006b).

countries that have seen rapid economic growth, notably East Asia, led by China. China alone accounted for 57 percent of the increase in total meat production in developing countries. For milk, developments are less spectacular but still remarkable: total milk production in developing countries expanded by 118 percent between 1980 and 2002, with 23 percent of that increase coming from one country, India.

This dramatic increase in demand for livestock products (a transition called the "livestock revolution" by Delgado *et al.*, 1999), is poised to continue for another 10 to 20 years before slowing down (Delgado *et al.*, 1999). A few developing countries, notably Brazil, China and India are emerging as world players as their strength as trading partners is growing rapidly (Steinfeld and Chilonda, 2005). These three countries account for almost two-thirds of total meat production in developing countries and for more than half of the milk (Table 1.6). They also account for close to three–quarters of the growth in milk and meat production in all developing countries.

There is a great deal of variation in the extent and character of livestock sector growth. China and East Asia have experienced the most impressive growth in consumption and production, first in meat and more recently also in dairy. The region will need to import increasing amounts of feed, and perhaps also livestock products, to meet future consumption growth. In contrast, India's livestock sector continues to be dairy-oriented, using traditional feed resources and crop residues. This picture is likely to change, as the booming poultry industry will pose feed demands that will far exceed current supplies. In contrast, Argentina, Brazil and other Latin American countries have successfully expanded their domestic feed base, taking advantage of low production costs and abundance of land (Steinfeld and Chilonda, 2006). They have moved to adding value to feed, rather than exporting it. They are poised to become the major meat-exporting region supplying developed and East Asian countries.

In the developing countries, livestock production is rapidly shifting towards monogastrics In fact, poultry and pigs account for 77 percent of the expansion in production. While total meat production in developing countries more than tripled between 1980 and 2004, the growth in ruminant production (cattle, sheep and goats) was only 111 percent, that of monogastrics expanded more than fourfold over the same period.

These dramatic developments in rapidly growing developing countries are in stark contrast with trends in developed countries, where consumption of livestock products is growing only slowly or stagnating. With low or no population growth, markets are saturated in most developed countries. Consumers are concerned about the health effects of high intake levels of livestock products, in particular red meat and animal fats. Continuous high-level consumption of these products is associated with a series of cardio-vascular diseases and certain types of cancer. Other perceived health problems associated with animal products sporadically and sometimes permanently suppress demand for animal products. These include the presence of residues (of antibiotics, pesticides, dioxins) and of pathogens (*Escherichia coli*, salmonella, mad cow disease).

In developed countries, total livestock production increased by only 22 percent between 1980 and 2004. Ruminant meat production actually declined by 7 percent while that of poultry and pigs increased by 42 percent. As a result, the share of production of poultry and pigs has gone up from 59 to 69 percent of total meat production. Among the monogastrics, poultry is the commodity with the highest growth rates across all regions. A main reason for this, apart from very favourable feed conversion, is the fact that poultry is a meat type acceptable to all major religious and cultural groups.

A few general observations can be made. The trend towards rapidly increasing livestock production in the tropics poses a series of technical problems, such as those related to climate and disease. Countries do not appear to be readily prepared for some of these, as has been demonstrated by the outbreaks of avian influenza in the last two years. The surge in production also entails an expansion of feed supplies and, particularly in Asia, an increasing amount will need to come from imports. Some countries will be faced with the question whether to meet this demand by importing feed for domestic livestock production or to opt for imports of livestock products. Production is also moving away from established production areas that have high environmental standards. This potentially creates opportunities for evading environmental controls.

On the consumption side, there is a trend towards global convergence of diets. Cultural peculiarities, though still strong in some areas, become increasingly blurred as demonstrated by the surge of poultry consumption in South and East Asia. This convergence is further driven by the fact that similar eating habits, such as fast and convenience food, are catching hold almost everywhere.

Most of the expansion in the supply of livestock products in developing countries comes from increased production, and only a relatively small part from imports. For developing countries as a whole, net imports account for only about 0.5 percent of total supply for meat, and 14.5 percent for milk (FAO, 2006b). However, trade in livestock products has increased much faster than trade in feed. For feedgrains, the traded share of total production has remained fairly stable in the range of 20 to 25 percent over the last decade. The share for meat increased from 6 percent in 1980 to 10 percent in 2002, and for milk from 9 to 12 percent over the same period.

Growth in trade in livestock products is also outpacing that of growth in production, facilitated by declining tariff barriers within the context of the General Agreement on Tariffs and Trade (GATT). This indicates a gradual trend towards producing livestock in locations where feed is available, rather than close to consumption centres - a trend made possible by infrastructure development and the establishment of refrigerated supply chains ("cold chains") in major producing countries.

Structural change

The large increases in supply of livestock products have been facilitated by structural adjustments in the sector, including growing intensities (discussed above), increasing scales of production, vertical integration and geographical shifts.

Units scale up in size, while smallholders are marginalized

There has been a rapid growth in the average size of primary production units, accompanied by a substantial decline in the numbers of livestock producers in many parts of the world. The major driver of this process is the cost reduction that can be realized through the expansion of scale of operations at various stages of the production process. Smallholders may stay in the livestock business by selling their products at prices that

A Maasai woman carrying a baby on her back milks a cow as its calf attempts to nurse. A gourd is used to collect the milk. The cattle are kept over night inside the perimeter of the boma to protect them from wildlife – Kenya 2003

value their own labour input below the market rate. However, this occurs mostly in countries with limited employment opportunities in other sectors. As soon as employment opportunities in other sectors arise, many smallholder producers opt out of livestock production.

Different commodities and different steps of the production process offer different potential for economies of scale. The potential tends to be high in post-harvest sectors (slaughterhouse, dairy plants). Poultry production is most easily mechanized, and industrial forms of production emerge even in least developed countries. In contrast, dairy production shows fewer economies of scale because of the typically high labour input. As a result, dairy production continues to be dominated by family-based production.

For dairy and small ruminant production, farm-level production costs at the smallholder level are often comparable with those of large-scale enterprises, usually because of the cost advantages of providing family labour below the level of the minimum wage. However, the expansion of smallholder production beyond a semi-subsistence level is constrained by a number of barriers, lack of competitiveness and risk factors (see below).

Access to land and credit is an increasing problem. Recent LEAD studies (Delgado and Narrod, 2006) show the substantial impact of hidden and overt subsidies that facilitate the supply of cheap animal products to the cities, to the disadvantage of small-scale rural producers. There is often no public support to adapt or disseminate new technologies for small-scale use. Production costs are higher at the smallholder level because of both market and production risks. Market risks include price fluctuations for both inputs and products. These are often amplified for smallholders because of their weak negotiating position. Some small-scale producers evolved from subsistence farming with sound risk coping mechanisms, but lack the assets or strategies to sustain full exposure to market risks. The absence of safety nets in the face of economic shocks, invariably present in such markets, restricts the participation of smallholders. Production risks relate to resource degradation, control of assets such as land and water, climatic variations such as droughts and floods, and infectious diseases.

Smallholders face additional problems because of the transaction costs involved in product marketing. These are often prohibitively high because of the small quantities of marketable product produced and the absence of adequate physical and market infrastructures in remote areas. Transaction costs are also increased where producers lack negotiating power or access to market information, and remain dependent on intermediaries. Moreover, the frequent absence of producers' associations or other partnership arrangements makes it more difficult for smallholder producers to reduce transaction costs through economies of scale.

The desire to reduce transaction costs is a main force promoting vertical integration in developed and developing countries alike. In developing countries, it is found particularly in poultry and pork, but also in dairy production. These economic forces are sometimes further strengthened if governments tax market transactions, for example for feed, as described by

Delgado and Narrod (2002) in the case of poultry producers in Andhra Pradesh (India). The combined effect of economic gains from lowering transaction costs by vertical integration, and more favourable tax regimes for larger enterprises, tends to disadvantage independent and small-scale producers severely.

Geographic shifts

Production grows more concentrated

Traditionally, livestock production was based on locally available feed resources, particularly those having limited or no alternative use value, such as natural pasture and crop residues. The distribution of ruminants was almost completely determined by the availability of such resources. The distribution of pigs and poultry followed closely that of humans, because of their role as waste converters. For example, a LEAD study in Viet Nam (a country in its early stages of industrialization) found that 90 percent of the poultry distribution pattern could be explained by the distribution of the human population (Tran Thi Dan *et al.*, 2003).

In the course of development, the livestock sector strives to free itself from local natural resource constraints – but becomes subject to a different set of factors that shape its geographical distribution and concentration. The importance of agro-ecological conditions as a determinant of location is replaced by factors such as opportunity cost of land and access to output and input markets.

As soon as urbanization and economic growth translate rising incomes into "bulk" demand for animal source food products, large-scale operators emerge. At the initial stage, these are located close to towns and cities. Livestock products are among the most perishable food products, and their conservation without chilling and processing poses serious quality and human health problems. Therefore, livestock have to be produced close to the location of demand, unless there is adequate infrastructure and technology to permit livestock to be kept farther away.

At a later stage, livestock production shifts even further from demand centres, driven by factors such as lower land and labour prices, access to feed, lower environmental standards, tax incentives or locations with fewer disease problems. The LEAD study found that the poultry density in areas closer than 100 km to Bangkok decreased between 1992 and 2000, with the largest decrease (40 percent) in the areas close to the city (less than 50 km). Density increased in all areas further away than 100 km (Gerber *et al.*, 2005).

The LEAD study found that for all countries analysed (Brazil, France, Mexico, Thailand, Viet Nam), despite the variety of factors that determine optimal location, there is a continuing process of concentration for all species covered by the analysis (cattle, chicken and pigs). Even in developed economies, the trend of concentration and increasing scale is continuing.

Vertical integration and the rise of supermarkets

Large multinational firms are becoming dominant in the meat and dairy trade, both in the developed world and in many developing countries experiencing fast livestock sector growth. Their strength is linked to achieving economies of size and scope, and to sourcing supplies at

Breeding sows in Rachaburi – Thailand 2004 different levels and across national boundaries. Vertical integration allows not only for gains from economies of scale. It also secures benefits from market ownership and from control over product quality and safety, by controlling the technical inputs and processes at all levels.

The rapid expansion of supermarkets and fast food outlets in developing countries started in the 1990s and has already large segments of the market in Latin America, East Asia and West Asia; this process has now also started in South Asia and sub-Saharan Africa. This expansion has been accompanied by a relative decline of traditional "wet" and local markets. For example, in China the number of supermarket outlets rose from 2 500 in 1994 to 32 000 in 2000 (Hu and Reardon, 2003). The supermarket share of total retail turnover is estimated to have reached about 20 percent of the total packaged and processed food retailing (Reardon *et al.*, 2003). According to the same authors, the share of supermarkets in the retailing of fresh foods is about 15 to 20 percent in Southeast Asia. India still has a comparatively low supermarket share of only 5 percent. As is already the case in developed countries, the large-scale retail sector is becoming the dominant actor in the agrifood system.

The rise of supermarkets has been facilitated by innovations in retail procurement logistics, technology and inventory management in the 1990s, with the use of the Internet and information management technology. This has enabled centralized procurement and consolidated distribution. The technological change, led by global chains, is now diffusing around the world through knowledge transfer, and imitation by domestic supermarket chains. The substantial savings from efficiency gains, economies of scale and coordination cost reduction provide profits for investment in new stores, and, through intense competition, reduce prices to consumers. The requirements of these integrated food chains, in terms of volume, quality, safety, etc. are becoming pervasive throughout the livestock sector.

In summary, the trends in the global livestock sector can be described as follows:

- Demand and production of livestock products are increasing rapidly in developing countries that have outpaced developed countries. A few large countries are taking centre stage. Poultry has the highest growth rate.
- This increasing demand is associated with important structural changes in countries' livestock sectors, such as intensification of production, vertical integration, geographic concentration and up-scaling of production units.
- There are concomitant shifts towards poultry and pig meat relative to ruminant meat, and towards grain- or concentrate-based diets relative to low-value feed.

These trends indicate a growing impact on the environment, as will be shown in more detail in the following chapters. Growth in itself may be regarded as a problem as it is not offset by concomitant productivity gains. Although these are important, the expanding livestock sector lays hands on additional feed and land resources that come at significant environmental costs. Structural change also modifies the nature of damage. In addition to issues associated with extensive production, such as overgrazing, there is a steep increase in those connected to intensive and industrial forms, such as concentration of pollutants, expansion of arable land for feed and environmental health problems. Further, the shift to traded and processed feeds spreads the environmental problems to other sectors, e.g. feedcrop production, fisheries, and to other parts of the world, which often obscures the real nature and extent of environmental impact.

02

Livestock in geographic transition

This chapter deals with the changing use of land[1] by livestock and some of the environmental impacts of that use[2]. Land management has a direct impact on the biophysical conditions of the land including soil, water, fauna and flora.

Land use has both spatial and temporal dimensions. Types of land use can spread or shrink, scatter or concentrate, while land use at a single location can be stable, seasonal, multiple and/or transitory. Land use is driven by a wide range of factors: some are endogenous to the land (e.g. bio-physical characteristics), some relate to the individual or the society using the land (e.g. capital availability, technical knowledge), some, finally, depend on the institutional and economic framework in which the land-user operates (e.g. national policies, markets, services).

Access to land and its resources is becoming an increasingly acute issue and source of competition among individuals, social groups and nations. Access to land has driven disputes

[1] With UNEP (2002), we define land as the terrestrial bioproductive system that comprises soil, vegetation - including crops, other biota, and the ecological and hydrological processes that operate within the system".

[2] Land-use changes include land cover-changes as well as the changing ways in which the land is managed. Agricultural land management refers to the practices by which humans use vegetation, water and soil to achieve a given objective. e.g. use of pesticides, mineral fertilizers, irrigation and machinery for crop production (Verburg, Chen and Veld Kamp, 2000).

and wars throughout history and, in some areas, resource-related conflicts are on the increase. For example, disputes over access to renewable resources, including land, are one of the principal pathways in which environmental issues lead to armed conflicts (Westing, Fox and Renner, 2001). This may be the result of a reduced supply of land (because of depletion or degradation), distribution inequities or a combination of these factors. Increasing land prices also reflect the increasing competition for land. (MAFF- UK, 1999).

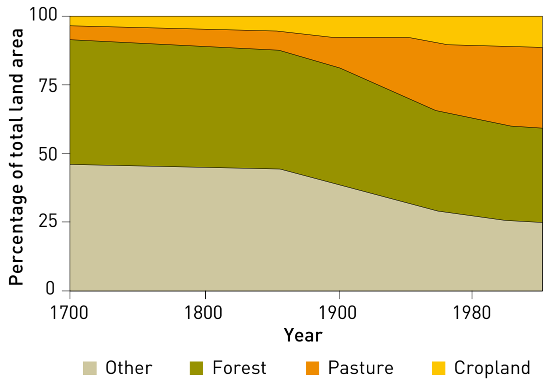
In this chapter, we will first look at the broad trends in land use and the forces that drive them, and introduce the "livestock transition" as a basic concept central to the understanding of livestock-environment interactions. We will then take a closer look at how the demand for livestock food products is distributed in relation to population and income. We will then turn to the geographic distribution of the natural resource base for livestock production, especially feed resources. This includes grazing land and arable land, particularly where surplus crop production is being used as feed for livestock production. Resources for livestock production and demand for animal products are balanced through livestock production systems that interact with both the resources and demand side. We will look at the changing geography of production systems and the way transport of feed and animal products resolve geographical mismatches and bring about different competitive advantages. Finally, we will review the main land degradation issues related to the livestock sector.

2.1 Trends in livestock-related land use

2.1.1 Overview: a regionally diverse pattern of change

The conversion of natural habitats to pastures and cropland has been rapid. Conversion accelerated after the 1850s (Goldewijk and Battjes, 1997) (Figure 2.1). More land was converted to crops between 1950 and 1980 than in the preceding 150 years (MEA, 2005a).

Figure 2.1 **Estimated changes in land use from 1700 to 1995**



Source: Goldewijk and Battjes (1997).

Table 2.1 presents regional trends over the past four decades for three classes of land use: arable land, pasture and forest. In North Africa, Asia, Latin America and the Caribbean land use for agriculture, both for arable and pasture, is expanding. The expansion of agriculture is fastest in Latin America and sub-Saharan Africa, mostly at the expense of forest cover (Wassenaar *et al.*, 2006). In Asia (mostly Southeast Asia) agriculture is expanding, and showing even a slight acceleration. In contrast, North Africa has seen crop, pasture and forestry expanding at only modest rates, with low shares of total land area covered by arable land. Oceania and sub-Saharan Africa have limited arable land (less than 7 percent of total land) and vast pasture land (35 to 50 percent of total land). Expansion of arable land has been substantial in Oceania and is accelerating in sub-Saharan Africa. There is a net reduction of forest land in both regions. Local studies have also found replacement of pasture by cropland. In sub-Saharan Africa, where cropping and grazing are often practised by different ethnic groups, the advance of crops into pasture land often results in conflict, as shown by major disturbances in the Senegal river basin between Mauritania and Senegal, and in North Eastern Kenya, between the Boran and the Somalis (Nori, Switzer and Crawford, 2005).

Box 2.1 Recent trends in forestry expansion

The Global Forest Resources Assessment 2005 suggests that forest still covers less than 4 billion hectares, or 30 percent of the total land surface area. This area has been in continuous decrease, although at a slowing pace. The net loss in forest area is estimated at 7.3 million hectares per year over the period 2000 to 2005, compared to 8.9 million hectares per year over the period 1990 to 2000. Plantation forests are generally increasing but still account for less than 4 percent of total forest area (FAO, 2005e). On average, 2.8 million hectares of forest were planted each year over the period 2000 to 2005.

These global figures mask differences among regions and forest types. Africa, North, Central and South America and Oceania showed net forest cover losses over the period 2000 to 2005 (FAO, 2005e), with the two latter bearing the largest losses. In contrast, forest cover increased in Asia over the same period, owing to large-scale reforestation in China, and continued to increase in Europe, although at a slowing pace. Primary forest area in Europe and Japan is expanding, thanks to strong protection measures.

Forest cover embraces a range of land usages. Wood production continues to be a major function in many forests. Trends are diverging though: Africa showed a steady increase in wood removal over the period 1990 to 2005, while production is decreasing in Asia. Forests are increasingly designated for the conservation of biodiversity. This kind of forest (mainly in protected areas) increased by an estimated 96 million hectares over the 1990 to 2005 period, and by 2005 accounted for 11 percent of all forests. Soil and water conservation is seen as a dominant function for 9 percent of the world's forests.

Source: FAO (2005e).

Western Europe, Eastern Europe and North America show a net decrease in agricultural land use over the last four decades, coupled with stabilization or increase in forest land. These trends occur in the context of a high share of land dedicated to crops: 37.7 percent, 21 percent and 11.8 percent in Eastern Europe, Western Europe and North America, respectively. The Baltic States and Commonwealth of Independent States (CIS) states show an entirely different pattern, with decreasing land dedicated to crops and increasing land dedicated to pasture. This trend is explained by economic regression causing the abandonment of cropland, and by structural and ownership changes that occurred during transition in the 1990s. Map 1 (Annex 1) further shows the uneven geographical distribution of cropland, with vast areas remaining mostly uncropped on all continents. The main patches of highly intensive cropping are found in North America, Europe, India and East Asia.

The massive expansion of arable and pasture land over the last four decades has started to slow (Table 2.1). At the same time, human populations grew more than six times faster, with annual growth rate estimated at 1.9 percent and 1.4 percent over the 1961–1991 and 1991–2001 periods respectively.

Extensification gives way to intensification

Most of the increase in food demand has been met by intensification of agricultural land use rather than by an expansion of the production area. The total supply of cereals increased by 46 percent over the last 24 years (1980 to 2004), while the area dedicated to cereal production shrank by 5.2 percent (see Figure 2.2). In developing countries as a whole, the expansion of harvested land accounted for only 29 percent of the growth in crop production over the period 1961–99, with the rest stemming from higher yields and cropping intensities. Sub-Saharan Africa,

Table 2.1

Regional trends in land use for arable land, pasture and forest from 1961 to 2001

	Arable land			Pasture			Forest		
	Annual growth rate (%)		Share of total land in 2001 (%)	Annual growth rate (%)		Share of total land in 2001 (%)	Annual growth rate (%)		Share of total land in 2002[2] (%)
	1961–1991	1991–2001		1961–1991	1991–2001		1961–1991	1990–2000[2]	
Developing Asia[1]	0.4	0.5	17.8	0.8	0.1	25.4	-0.3	-0.1	20.5
Oceania	1.3	0.8	6.2	-0.1	-0.3	49.4	0.0	-0.1	24.5
Baltic states and CIS	-0.2	-0.8	9.4	0.3	0.1	15.0	n.d.	0.0	38.3
Eastern Europe	-0.3	-0.4	37.7	0.1	-0.5	17.1	0.2	0.1	30.7
Western Europe	-0.4	-0.4	21.0	-0.5	-0.2	16.6	0.4	0.4	36.0
North Africa	0.4	0.3	4.1	0.0	0.2	12.3	0.6	1.7	1.8
Sub-Saharan Africa	0.6	0.9	6.7	0.0	-0.1	34.7	-0.1	-0.5	27.0
North America	0.1	-0.5	11.8	-0.3	-0.2	13.3	0.0	0.0	32.6
Latin America and the Caribbean	1.1	0.9	7.4	0.6	0.3	30.5	-0.1	-0.3	47.0
Developed countries	0.0	-0.5	11.2	-0.1	0.1	21.8	0.1	n.d.	n.d.
Developing countries	0.5	0.6	10.4	0.5	0.3	30.1	-0.1	n.d.	n.d.
World	0.3	0.1	10.8	0.3	0.2	26.6	0.0	-0.1	30.5

[1] Data on pasture excludes Saudi Arabia.
[2] Data for 2000 obtained from FAO, 2005e.
Note: n.d. - no data.
Source: FAO (2005e; 2006b).

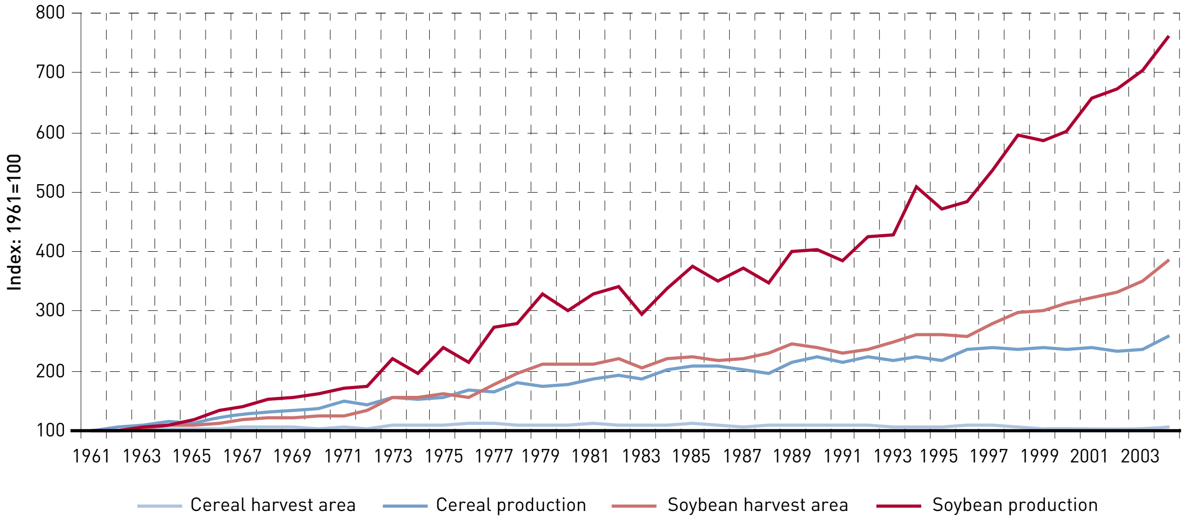
where area expansion accounted for two-thirds of growth in production, was an exception.

The intensification process has been driven by a range of factors (Pingali and Heisey, 1999). In Asia, where extraordinary growth in cereal productivity has been achieved, rising land values owing to increasing land scarcity have been the dominant factor. Cereal yields have also substantially increased in some Latin American and African countries. With lower population densities than in Asia, the forces influencing intensification have been the level of investments in market and transport infrastructure and the extent to which countries engaged in export-oriented trade. In contrast, productivity gains were low in sub-Saharan Africa, despite population growth. Relative land abundance (in comparison to Asia), poor market infrastructure and lack of capital contributed to the modest performance.

Technically, increased productivity can be achieved by increased cropping intensity (i.e. multiple cropping and shorter fallow periods) by higher yields, or by a combination of the two. Higher yields are the result of technological advances and higher input use in crop production – notably irrigation, modern high-yielding plant varieties, fertilizers and mechanization. Use of tractors, mineral fertilizers and irrigation increased strongly between 1961 and 1991, and much more slowly afterwards (see Table 1, Annex 2). In comparison, use of mineral fertilizers has substantially decreased since 1991 in developed countries, as a result of more efficient resource use and environmental regulations aimed at reducing nutrient loading.

While scope for productivity increases still exists, Pingali and Heisey (1999) show that the productivity of wheat and rice in lowland Asia has lately been growing at a dwindling pace. Key factors explaining this slowing trend are land degradation, declining research and infrastructure investment, and increasing opportunity cost

Figure 2.2 **Total harvested area and total production for cereals and soybeans**



Source: FAO (2006b).

of labour, although new technological developments (i.e. hybrid rice) might enable new growth. Arable land expansion will likely continue to be a contributing factor in increasing agricultural production. In particular, this will be the case for developing countries, where arable land expansion, increases in cropping intensity and yield increases accounted for 23, 6 and 71 percent, respectively, of crop production growth over the 1961 to 1999 period, and they are expected to account for 21, 12, and 67 percent, respectively over the 1997/99 to 2030 period (FAO, 2003a). In developed countries, in contrast, the increase in production is expected to be reached with a constant or locally declining arable area. The foreseen shift to biofuels, and the increased demand for biomass that will result may, however, lead to a new area of crop expansion, especially in Western Europe and North America.

2.1.2 Globalization drives national land-use changes

Changes in agricultural land use are driven by a wide range of factors. Ecological conditions, human population density and level of economic development provide the broad context of land use, along with more localized factors specific to each area. Individual and social decisions leading to land changes are also increasingly influenced by changing economic conditions and institutional frameworks (Lambin *et al*., 2001).

Two concepts are central in explaining agricultural land-use changes: profit per unit of land and opportunity cost. Profit per unit of land[3] describes the potential interest for an operator to engage in a particular use of the land. Profit generally depends on the biophysical characteristics of the land, on its price, and factors including accessibility to markets, inputs and services. On the other hand, the opportunity cost[4] compares the economic and social costs of two or more ways of using the same piece of land. Opportunity cost includes not only the private costs of production, but also direct and indirect costs borne by society, such as losses

[3] The surplus of revenue generated over expenses incurred for a particular period of time.

[4] Opportunity cost can be defined as the cost of doing an activity instead of doing something else.

of ecosystem services. For example, part of the opportunity cost of cropping an area would be loss of the possibility of using it for recreational purposes.

In a context where non-marketable ecosystem services are not priced, decisions on land use are predominantly driven by calculation of private profit per unit of land, usually based on tradable goods and services. As a result, the non-marketed benefits are often lost, or external costs are imposed on society. However, the environmental and social services provided by ecosystems are receiving increasing recognition.

A case in point is the growing recognition of the wide range of services provided by forest, a type of land use generally antagonistic to agricultural use, although modern agroforestry technologies do produce some synergies. Forests are increasingly used for conservation of biodiversity (see Box 2.1). This is a global trend, although the pace is significantly slower in Oceania and Africa.

Soil and water conservation is also seen as a dominant function for 9 percent of the world's forests. Recreation and education activities are another use of forest that is on the increase and represent the primary management objective for 2.4 percent of the forest in Europe, while 72 percent of the total forest area was acknowledged to provide social services (MEA, 2005a).

Roundwood removal, on which the calculation of profit per unit of forest land is usually based, was estimated at US$64 billion worldwide in 2005. This has been decreasing in real terms over the last 15 years (FAO, 2005e). In a case study of the economic value of forest in eight Mediterranean countries, non-wood forest products, recreation, hunting, watershed protection, carbon sequestration and passive use accounted for 25 to 96 percent of the total economic value of the forests. Non-marketed economic values (e.g. watershed protection, carbon sequestration, recreation, non-timber forest products) were estimated higher than commonly measured economic values (e.g. grazing, timber and fuelwood) in three countries (Italy, Portugal and the Syria Arab Republic), though they were lower in five (Algeria, Croatia, Morocco, Tunisia and Turkey) (MEA, 2005a).

As economies continue to liberalize, local agricultural goods compete with equivalent goods produced farther away. Increasingly, therefore, agricultural land-use opportunities are competing across continents. Both profit per unit of land and opportunity costs of agricultural land use vary immensely around the globe, depending on agro-ecological conditions, access to markets, availability of production inputs (including services), existence of competitive land usage and valuation of ecosystem services. Agricultural production relocates accordingly, resulting in changes in use of agricultural land and also of forests and other natural areas. For example, New Zealand lamb competes with local produce in Mediterranean markets. New Zealand lamb is produced at a relatively low cost because of much lower opportunity cost of land (mainly owing to a much lower recreational demand) and higher productivity of pasture. As a result, the marginal pastures traditionally used for sheep production in the EU Mediterranean basin are progressively being abandoned to natural vegetation and other recreational usages.

The process through which former agricultural land reverts to forest has been called the "forest transition". Mainly, the term has been applied to developed countries in Europe and North America (Mather, 1990; Walker, 1993; Rudel, 1998).

During the early period of colonization and economic growth, settlers and farmers cleared land rapidly to provide agricultural goods required by local populations. Later, as urban development came to dominate and trade expanded, rural populations moved to cities, and agricultural markets traded with increasingly distant locations of demand and supply. There were huge gains in agricultural productivity in areas with high agricultural potential.

This resulted in substantial land-use shifts: farming moved into the remaining unused fertile lands, and marginal locations were abandoned, especially in remote areas with poor soil characteristics. More productive land with good accessibility remained in production. As abandoned land reverted to natural vegetation cover, this led to net reforestation in parts of Europe and North America, from the end of the nineteenth century on (Rudel, 1998). Forest transition is an ongoing trend in Europe and North Africa and has shown similar patterns in Asia, although national policies may have fostered the process in the latter (Rudel, Bakes and Machinguiashi, 2002). Map 2 (Annex 1) shows areas of net forest area gain in the USA, Southern Brazil, Europe and Japan.

2.1.3 Land degradation: a vast and costly loss

Land degradation is widely recognized as a global problem having implications for agronomic productivity and the environment as well as effects on food security and quality of life (Eswaran, Lal and Reich, 2001). Although the magnitude of the problem is broadly shared, there are a number of definitions for land degradation, interpreted in different ways among various disciplinary groups. We here refer to the definition of the United Nations Environment Programme (UNEP) where "Land degradation implies a reduction of resources potential by one or a combination of processes acting on the land, such as: (i) soil erosion by wind and/or water, (ii) deterioration of the physical, chemical and biological or economic properties of soil; and (iii) long-term loss of natural vegetation" (UNEP, 2002).

Agricultural land degradation is of particular concern on several grounds, as it reduces productivity, which in turn leads to further expansion of agricultural land into natural habitats. It also requires additional natural resources to restore the land (e.g. lime to neutralize acidity, water to flush out salinity), and can generate pollution with off-site impacts (Gretton and Salma, 1996). Intensification and extensive land use can both result in environmental impacts, though in different ways. Intensification has both positive and negative effects. Increased yields in agricultural systems help to reduce the pressure to convert natural ecosystems into cropland, and can even allow for re-conversion of agricultural land back to natural areas, as observed in OECD countries.

However, the increased inputs of fertilizers, biocides and energy that intensification involves have also increased pressure on inland-water ecosystems, generally reduced biodiversity within agricultural landscapes and generated more gaseous emissions from higher energy and mineral fertilizers inputs (MEA, 2005a). On the other hand, extensive use of land for pasture or cropping has also often led to the deterioration of vegetative cover and soil characteristics.

Environmental implications of land degradation are multiple. Among the most critical issues are the erosion of biodiversity (through habitat destruction or pollution of aquifers), climate change (through deforestation and the loss of soil organic matter releasing carbon to the atmosphere) and depletion of water resources (through alteration of the soil texture or removal of vegetation cover affecting water cycles). These mechanisms and their significance will be described in detail in the following chapters.

The differences in definitions and terminology for land degradation are responsible for the variations between results from studies that have attempted to evaluate the extent and rate of this process. Oldeman (1994) produced one of the generally accepted estimates of the extent of global land degradation. It estimates that about 19.6 million km^2 are degraded, mostly because of water erosion (Table 2.2). This figure does not, however, include loss of natural vegetation and, based on the above UNEP definition, is therefore more an estimate of soil degradation rather than of land degradation. Still, according to Oldeman (1994), about one-third of the land used as forests and woodlands appears to be degraded in Asia (ca 3.5 million km^2), against 15 to 20 percent in Latin America and Africa. Land degradation of pasture is mainly an issue in Africa

(2.4 million km^2), although Asia, and to a lesser extent Latin America are also affected (2.0 and 1.1 million km^2 respectively). Finally, about one-third of the agricultural land is degraded in Asia (2.0 million km^2), against half in Latin America, and two-thirds in Africa.

Desertification is a form of land degradation, taking place in arid, semi-arid and dry subhumid areas and resulting from various factors, including climatic variations and human activities (UNEP, 2002). Dregne and Chou (1994) estimated that degraded lands in dry areas of the world amount to 3.6 billion hectares or 70 percent of the 5.2 billion hectares of the total land areas considered in these regions (Table 2.3). These figures include loss of vegetal cover and are not directly comparable with the previous ones. Reich *et al.* (1999) further estimate that in Africa, about 6.1 million km^2 of land are under low to moderate degradation risk and 7.5 million km^2 are under high and very high risk. Cumulatively, desertification is estimated to affect about 500 million Africans, seriously undermining agricultural productivity despite good soil resources.

Yield reduction is one of the most evident economic impacts related to land degradation. In Africa, it is estimated that past soil erosion may have depressed yields by 2 to 40 percent, with a mean loss of 8.2 percent for the continent (Lal, 1995). In South Asia, water erosion is estimated to reduce harvests by 36 million tonnes of cereal equivalent every year, valued at US$5 400 million, while water erosion would cause losses estimated at US$1 800 million (FAO/UNDP/UNEP, 1994). Worldwide, it is estimated that 75 billion tonnes of soil are lost every year, costing approximately US$400 billion per year, or about US$70 per person per year (Lal, 1998). Analysis conducted at the International Food Policy Research Institute (IFPRI) (Scherr and Yadav, 1996) suggest that a slight increase in land degradation relative to current trends could result in 17–30 percent higher world prices for key food commodities in 2020, and increased child malnutrition. Besides diminishing food production and food security, land degradation hampers agricultural income and thereby economic growth, as shown by analysis supported by country models in Nicaragua and Ghana (Scherr and Yadav, 1996). Land degradation can ultimately result in emigration and depopulation of degraded areas (Requier-Desjardins and Bied-Charreton, 2006).

Long-term effects of land degradation and, in particular, the reversibility of land degradation processes and the resilience of ecosystems are subject to debate. Soil compaction, for example, is a problem in vast areas of cropland worldwide. It is estimated to be responsible for yield

Table 2.2

Estimates of the global extent of land degradation

Type	Light	Moderate	Strong + Extreme	Total
	(.................... million km^2 ....)			
Water erosion	3.43	5.27	2.24	10.94
Wind erosion	2.69	2.54	0.26	5.49
Chemical degradation	0.93	1.03	0.43	2.39
Physical degradation	0.44	0.27	0.12	0.83
Total	**7.49**	**9.11**	**3.05**	**19.65**

Source: Oldeman (1994).

Table 2.3

Estimates of all degraded lands in dry areas

Continent	Total area	Degraded area[1]	Percentage degraded
	(million km^2)	*(million km^2)*	
Africa	14.326	10.458	73
Asia	18.814	13.417	71
Australia and the Pacific	7.012	3.759	54
Europe	1.456	0.943	65
North America	5.782	4.286	74
South America	4.207	3.058	73
Total	**51.597**	**35.922**	**70**

[1] Comprises land and vegetation.
Source: Dregne and Chou (1994).

reductions of 25 to 50 percent in parts of the EU and North America, with on-farm losses estimated at US$1.2 billion per year in the United States. Compaction is also an issue in Western Africa and Asia (Eswaran, Lal and Reich, 2001). Soil compaction is, however, relatively easily reversed by adapting ploughing depth. Water and wind erosion, in contrast, have irreversible consequences, for example mobile sand dunes (Dregne, 2002). Reversal of the land degradation process often requires substantial investments, which may fall beyond investment capacity or do not grant satisfactory returns under current economic conditions. Rehabilitation costs of degraded land were estimated on average at US$40/ha/year for pastures, US$400/ha/year for rainfed cropland and US$4 000/ha/year for irrigated cropland in sub-Saharan Africa, with average investment periods of three years (Requier-Desjardins and Bied-Charreton, 2006).

2.1.4 Livestock and land use: the "geographical transition"

Historically, people raised livestock as a means to produce food, directly as meat and dairy products and indirectly as draught power and manure for crop production. Since conservation technology and transport facilities were poor, goods and services from livestock were used locally. Livestock were kept geographically close to human settlements, in most cases while pastoralists grazed animals on their migrations.

Distribution trends have varied according to the type of species. Monogastric species (e.g. pigs and poultry) have predominantly been closely associated with human populations, in household backyards. The reason is that monogastric species depend on humans for feed (e.g. household waste, crop by-products) and for protection from predators. The distribution of human populations and monogastric species is still closely correlated in countries with traditional production systems (FAO, 2006c; Gerber *et al.*, 2005). In the distribution of ruminant species (e.g. cattle, buffaloes, sheep, goats) feed and especially fodder resources have played an important role. The land area used for ruminant production is generally substantial. Ruminants have been herded where there are pasture resources, and only in exceptional cases have they been fed with harvested feed (e.g. draught animals or seasonally in cold areas). Herding ruminants involves daily or seasonal movements, over distances varying from hundreds of metres up to hundreds of kilometres in the case of large-scale transhumance or nomadism. Some or all of the humans relying on the herd are involved in the movement, sometimes keeping a geographical anchor area (e.g. village, *boma, territoire d'attache*).

In modern times, livestock production has developed from a resource-driven activity into one led mainly by demand. Traditional livestock production was based on the availability of local feed resources, in areas where disease constraints allowed this.

Modern livestock production is essentially driven by demand for livestock products (Delgado *et al.*, 1999), drawing on additional feed resources as required. As a result, the location of livestock production is undergoing important shifts. With the emergence of large economies such as China and India as new centres of demand and production (Steinfeld and Chilonda, 2006) these geographic shifts have accelerated globally over the last decades. The geography of livestock production and its changes are the keys to understanding livestock-environment interactions. For example, livestock waste does not pose an environmental problem in areas of sparse livestock density; on the contrary, it is a valuable input to crop activities and helps to maintain soil fertility. In contrast, in areas of high livestock density, the capacity of surrounding land or waters to absorb the waste is often exceeded and environmental damage ensues.

Access to markets, feed resources, infrastructure, prices for land, labour and transport and disease status affect the location of livestock production. In this chapter we will analyse the trends in livestock geography and the underlying determinants, to help understand and interpret the environmental consequences. We will examine first the overall extent of land devoted directly or indirectly to livestock production, and then the geographical distribution of the main stages and types of livestock production.

Land use intensification in the feed sector

The first main feature is livestock's demand for pasture and cropland, and the very substantial changes in area that have occurred in the past and continue to occur. Grazing land has expanded by a factor of six since 1800, and now covers roughly 35 million km^2, including large areas of continents where previously there had been little or no livestock grazing (North America, South America, Australia). In many areas, grazing has expanded to occupy virtually all the land that can be grazed and for which there is no other demand (Asner *et al.*, 2004). South America, Southeast Asia and Central Africa are the only parts of the world where there are still significant areas of forest that could be turned into grazing, but in the latter major investments in disease control would be needed. As described in Section 2.5, the expansion of pasture into forest ecosystems has dramatic environmental consequences.

More recent is the advent of grain-feeding to livestock, starting in the 1950s in North America, extending into Europe, the former Soviet Union and Japan in the 1960s and 1970s, now commonplace in much of East Asia, Latin America and West Asia. Grain-feeding is not widespread yet in most of sub-Saharan Africa and South Asia, but is rapidly increasing from a low base. This demand for feedgrains and other feed materials has greatly increased the arable land requirements of livestock production, from a very small area to about 34 percent of the total arable land today (see Section 2.3).

Both the long-term expansion of grazing land and the more recent expansion of arable land for feed will probably reach a maximum, followed by a future decrease. World population is expected on the UN's medium projection to grow to just over 9 billion in 2050, about 40 percent more than today, and to begin decreasing shortly thereafter (UN, 2005). Population growth will combine with changes in incomes and urbanization rates to determine global trends in demand for animal-derived food, though the details are, of course, uncertain. In some developed countries, demand growth is already slowing or declining. In emerging economies, the ongoing livestock revolution is also poised for a slow down, as the tremendous increases in per capita livestock consumption of the past two decades have already occurred, and population growth continues to slow.

In fact, growth rates of livestock production for all developing countries peaked in the 1990s at 5 percent per year, falling to an average of 3.5 percent for the 2001–2005 period. In Asia and the Pacific, where China drove the livestock revolution, average annual growth rates peaked in the 1980s at 6.4 percent, and have decreased since then to 6.1 percent in the 1990s and 4.1 percent over the 2001 to 2005 period. Production followed a similar pattern in West Asia and North Africa. Some regions may, however, not yet have reached their peak in production growth. Growth rates patterns are less clear in Latin America and may well further increase, pulled by export-oriented production in countries such as Argentina or Brazil. Consumption and production are still very low in Africa and will increase as economic growth allows. Finally, production growth is expected to be strong in transition countries, recovering to previous levels. Despite these areas of expansion, it is probable that the bulk of global growth in livestock production has already occurred and that further growth will take place at diminishing rates.

At the same time, intensification and the continued shift from ruminants to monogastrics (especially poultry) are continuously improving land-use efficiency, helping to reduce the land area used per unit of output. This is reinforced by the effect of increased feedcrop production efficiency, demonstrated by the continuing yield increases in all major feedcrops described above. By reducing post-harvest losses, advances in processing and distribution technology and practices also reduce the land required per unit of consumed products. The combined effect, in many developed countries, has been a decrease in the extent of grazing land, amounting to, for example, 20 percent since 1950 in the United States.

Two antagonist trends are thus at play: on the one hand production growth will further increase land demand by the sector, though at diminishing growth rates. On the other, continuous intensification will reduce the area of land used per unit of output. The relative strength of these two trends will determine the trend in total area used by livestock. It is suggested here that the global land requirements of the livestock sector will soon reach a maximum and then decrease. Grazing areas will start to decline first, followed by a reduction in land required for feed production. This overall trend is proposed as a model for understanding of livestock geography dynamics.

Locations shift in relation to markets and feed sources

The second major feature in livestock geography is livestock's changing spatial distribution: the geographical association with the feed base on the one hand, and with people and their needs for animal products on the other. At pre-industrial levels of development, monogastrics and ruminants follow different patterns of distribution. The distribution of monogastrics follows that of human settlements. When humans live predominantly in rural areas, so do monogastrics. In the early phases of industrialization, occurring today in many developing countries, humans rapidly urbanize, and so do monogastrics, usually in a peri-urban belt around consumption centres. This rural to peri-urban shift creates significant environmental problems and public health hazards. In a third phase, these problems are corrected by the gradual relocation of farms farther away from cities, once living standards, environmental awareness and institutional capacity permit. The same pattern applies for ruminants, but is less pronounced because their higher daily fibre requirements entail bulk movement of fodder, and the cost of this acts as a brake to the urbanization of livestock. Ruminant production, both meat and milk, tends to be much more rural-based throughout the different phases of development, even though important exceptions exist (for example, peri-urban milk production, such as observed in India, Pakistan and around most sub-Saharan cities).

The rapid urbanization of livestock, in particular monogastrics, and the subsequent gradual de-urbanization is a second distinct pattern taking place alongside the land-use intensification of the sector. Both patterns have immense implications for livestock's impact on the environment, and constitute the basic theme of this and the following chapters. We will use the expression "*livestock transition*" as a short form for these two patterns.

2.2 Geography of demand

On a global scale, the geographical distribution of the demand for animal-derived foods broadly follows that of human populations (Map 3, Annex 1). However, people have quite different demand patterns, depending on income and preferences. The rationale on which people select their food is complex, based on a number of objectives, and decisions are influenced by individual and societal capacity and preferences, as well as availability. Food preferences are undergoing rapid changes. While growing incomes in developing countries are increasing the intake of proteins and fats, some higher income segments in developed countries are cutting down on these components, for a number of reasons including health, ethics and an altered trust in the sector. On average, per capita consumption of animal-derived foods is highest among high-income groups, and growing fastest among lower- and middle-income groups in countries experiencing strong economic growth. The first group is

Table 2.4

Livestock and total dietary protein supply in 1980 and 2002

	Total protein supply from livestock		Total protein supply	
	1980	2002	1980	2002
	(.............. g/person ....)			
Sub-Saharan Africa	10.4	9.3	53.9	55.1
Near East	18.2	18.1	76.3	80.5
Latin America and the Caribbean	27.5	34.1	69.8	77.0
Asia developing	7.0	16.2	53.4	68.9
Industrialized countries	50.8	56.1	95.8	106.4
World	20.0	24.3	66.9	75.3

Source: FAO (2006b).

mostly concentrated in OECD countries, while the latter is mostly located in rapidly growing economies, such as Southeast Asia, the coastal provinces of Brazil, China and parts of India. The two groups coincide geographically in urban centres in rapidly growing economies.

Table 2.4 provides an overview of the important changes that have occurred in the average protein intake of people in various world regions. People in industrialized countries currently derive more than 40 percent of their dietary protein intake from food of livestock origin (the figures do not include fish and other seafood), and little change has occurred between 1980 and 2002. Changes have been most dramatic in developing Asia, where total protein supply from livestock for human diets increased by 140 percent, followed by Latin America where per capita animal protein intake rose by 32 percent. In contrast, there has been a decline in consumption in sub-Saharan Africa, reflecting economic stagnation and a decline in incomes. Detailed consumption patterns are shown in Table 2, Annex 2. The increasing share of livestock products in the human diet in many developing countries is part of a dietary transition that has also included a higher intake of fats, fish, vegetables and fruit, at the expense of staple foods, such as cereals and tubers.

Two major features emerge from these trends. First, there is the creation of new growth poles in emerging economies, with Brazil, China and India now being global players. Meat production in the developing countries overtook that of developed countries around 1996. Their share of production is projected to rise to about two-thirds by the year 2030 (FAO, 2003a). In contrast, in developed countries both production and consumption are stagnating and in some places declining. Second is the development of demand hotspots – urban centres - with high consumption per capita, fast aggregate demand growth, and a shift towards more processed animal-derived foods. A certain homogenization of consumed products (e.g. chicken meat) is also observed, although local cultures still have strong influence.

2.3 Geography of livestock resources

Different livestock species have the capacity to utilize a wide variety of vegetative material. Usually, feedstuff is differentiated into roughage, such as grass from pastures and crop residues, and feed concentrates, such as grains or oilseeds. Household waste and agro-industrial by-products can also represent a large share of feed resources.

2.3.1 Pastures and fodder

Variations in conversion, management and productivity

Grasslands currently occupy around 40 percent of the total land area of the world (FAO, 2005a; White, Murray and Rohweder, 2000). Map 4 (Annex 1) shows the wide distribution of pastures. Except in bare areas (dry or cold deserts) and dense forest, pastures are present to some extent in all regions. They are dominant in Oceania (58 percent of the total area – 63 percent in Australia), whereas their spread is relatively limited in West Asia and North Africa (14 percent) and South Asia (15 percent). In terms of area, four regions have 7 million km^2 of grassland or more: North America, sub-Saharan Africa, Latin America and the Caribbean and the Commonwealth of Independent States (see Table 3, Annex 2).

As Table 2.5 shows, grasslands are increasingly fragmented and encroached upon by cropland and urban areas (White *et al.*, 2000). Agriculture expansion, urbanization, industrial development, overgrazing and wildfires are the main factors leading to the reduction and degradation of grasslands that traditionally hosted extensive livestock production. The ecological effects of this conversion, on ecosystems, soil structure and water resources, can be substantial. There are, however, signs of an increasing appreciation of grassland ecosystems and the services they provide, such as biodiversity conservation, climate change mitigation, desertification prevention and recreation.

Permanent pastures are a type of human land use of grasslands, and are estimated to cover about 34.8 million km^2, or 26 percent of the total land area (FAO, 2006b). Management of pasture and harvested biomass for livestock varies greatly. On balance, although accurate estimates are difficult to make, biomass productivity of pastures is generally much lower than that of cultivated areas. A number of factors contribute to this trend. First, large pastures mainly occur in areas with marginal conditions for crop production (either temperature limited or moisture limited), which explain their low productivity in comparison to cropland. Second, in the arid and semi-arid rangelands, which form the majority of the world's grassland, intensification of the areas used as pasture is often technically and socio-economically difficult and unprofitable. Most of these areas already produce at their maximum potential. In addition, in much of Africa and Asia, pastures are traditionally common property areas that, as internal group discipline in the management of these areas eroded, became open access areas (see Box 2.2). Under such conditions any individual investor cannot capture the investments made and total investments levels will remain below the social optimum. Lack of infrastructure in these remote areas further contributes to the difficulty of successfully improving productivity through individual investments. In extensive systems, natural grasslands are thus only moderately managed.

However, where individual ownership prevails or traditional management and access rules are operative, their use is often carefully planned, adjusting grazing pressure seasonally, and mixing different livestock classes (e.g. breeding stock, young stock, milking stock, fattening stock) so as to reduce the risks of climate variability. In addition, techniques such as controlled burning and bush removal are practices that can

Table 2.5

Estimated remaining and converted grasslands

Continent and region	Percentage of				
	Remaining in grasslands	Converted to croplands	Converted to urban areas	Converted to other (e.g. forest)	Total converted
North America tallgrass prairies in the United States	9.4	71.2	18.7	0.7	90.6
South America cerrado woodland and savanna in Brazil, Paraguat and Bolivia	21.0	71.0	5.0	3.0	79.0
Asia Daurian Steppe in Mongolia, Russia and China	71.7	19.9	1.5	6.9	28.3
Africa Central and Eastern Mopane and Miombo in United Republic of Tanzania, Rwanda, Burundi, Dem. Rep. Congo, Zambia, Botswana, Zimbabwe and Mozambique.	73.3	19.1	0.4	7.2	26.7
Oceania Southwest Australian shrub lands and woodlands	56.7	37.2	1.8	4.4	43.4

Source: White, Murray and Rohweder (2000).

Box 2.2 The complex and weakening control of access to pastureland

Pastureland falls under a variety of property and access rights. Three types of land tenure are generally recognized, namely, private (an individual or a company), communal (a local community) and public (the state). Access rights can overlap with property rights, sometime resulting in a complex set of rules controlling the use of resources. Such discrepancies between access rules and the multiplicity of institutions responsible for their application often lead to conflicts among stakeholders claiming access to pastureland. In this regard, the Rural Code of Niger is an exemplar attempt to secure pastoralists' access to rangelands while maintaining such areas under a common property regime. The table below provides an overview of these rules and of the relative level of security they provide for the livestock keeper accessing the land resource. Access to water often adds another layer of access rights: in the dry lands, water plays a critical role as location of water resources are determinant to the use of pastures. Consequently, water rights are key to the actual access to arid and semi-arid pastureland. Holding no formal rights over land, pastoralists often do not get rights over water thereby suffering from a double disadvantage (Hodgson, 2004).

Table 2.6

Land ownership and access rights on pastoral land: possible combinations and resulting level of access security for the livestock keeper

<table>
<tr><th></th><th>No overlapping access right</th><th>Lease</th><th>Customary access rights[1]</th><th>Illegal intrusion or uncontrolled access</th></tr>
<tr><td>Private</td><td>+++
Freehold</td><td rowspan="2">from ++ to +++
Depends on the duration of the leasing contract and the strength of the institution that guarantees the leasing contract.</td><td>0 to ++
Issues may arise from the conflicting overlap between customary access right and recent land titling policies.</td><td>0 to +
Conflict</td></tr>
<tr><td>Communal</td><td>+++
Case of commonly/nationally owned herds</td><td>+ to +++
Customary access rights tend to loose strength and stability because of migrations and overlap with exogenous property and access right.</td><td>+ to ++
Depends on the relative strength of local communities/public administration and livestock keepers</td></tr>
</table>

Note: Level of stability in the access to the resource, from very high (+++) to very low (0)

[1] Customary access rights can take numerous forms. A common trait is their indentification of first and latecomers. They are thus quite vulnerable to strong migration fluxes, in which context they may exacerbate ethnic quarrels

Source: Chauveau, 2000; Médard 1998; Klopp, 2002.

improve pasture productivity, although they may also increase soil erosion and reduce tree and shrub cover. The low management level of extensive pasture is a major reason why such grasslands can provide a high level of environmental services such as biodiversity conservation.

For the purpose of this assessment, grasslands are grouped into three categories: extensive grasslands in marginal areas, extensive grasslands in high potential areas, and intensive pastures.

Box 2.2 (cont.)

Stability and security in accessing the pastoral resource are of utmost importance, as they are determinant to the management strategy the user will adopt. In particular, investments in practices and infrastructures improving pasture productivity may only be implemented if there is a sufficiently high probability to realize economic returns on the mid to long term. More recently, the existence of clear usage rights has shown to be indispensable to the attribution and remuneration of environmental services.

Table 2.7

Land use and land ownership in the United States

Acre	Cropland	Pasture	Forest	Other	Total
Federal	0	146	249	256	651
State and local	3	41	78	73	195
Indian	2	33	13	5	53
Private	455	371	397	141	1 364
Total	460	591	737	475	2 263
Relative percentages					
Federal	0	25	34	54	29
State and local	1	7	11	15	9
Indian	0	6	2	1	2
Private	99	63	54	30	60

Source: Anderson and Magleby (1997).

While detailed statistics are lacking, it is probably safe to say that most pasture land is private property, not common property and government land. Pasture are predominantly established on communal and public land in Africa (e.g. freehold land covers only about 5 percent of the land area in Botswana), South Asia (e.g. Commons, dominantly under pasture, account for around 20 percent of India's total land area), West Asia, China as well as Central Asia and Andean highlands. Furthermore, in Australia, most of the Crown Land - representing about 50 percent of the countries' area - is grazed under leases. In contrast, the majority of pasture land is titled under private ownership in Latin America and in the United States. Indeed, a survey on the United States shows that 63 percent of pastures are privately owned, while 25 percent belong to the Federal State and the rest to states and local communities (see Table 2.7). Finally, in Europe, pasture located in fertile low lands are generally privately owned, while marginal areas such as mountain rangelands and wetlands are usually public or communal, with traditional access rights.

Extensive pasture in marginal areas are defined here as having a net primary productivity of less than 1 200 grams of carbon per m^2/yr (Map 4, Annex 1; Table 4, Annex 2). This is the largest category by area (60 percent of all pastures), and is located mostly in dry lands and cold lands. This category is particularly dominant in developed countries, where it represents almost 80 percent of grasslands, while in developing countries it accounts for just under 50 percent of pastures. The contrast can be explained by differences in the opportunity cost of land: in developed countries, areas with good agro-ecological potential are generally used in more intensive forms than pasture. Grasslands in marginal areas are used extensively, either by mobile production systems (Africa, the CIS, South Asia and East Asia), or in large ranches (Oceania, North America). Using actual evapotranspiration (AET) as an indicator of vegetation climatic stress, Asner *et al.* (2004) show that in dry land biomes grazing systems tend to occupy the driest and climatologically most unstable regions, and in temperate biomes the most humid and/or cold parts. In terms of soils, the authors also show that grazing systems generally occupy the least fertile soils in the dry lands and the unfrozen soils of the boreal areas, along with both least fertile and moderately fertile soils in the tropical biomes. They conclude that the land frontier for further pasture expansion into marginal areas is exhausted.

Extensive pasture in high potential areas is defined as those with a net primary productivity of more than 1 200 g of carbon per m^2/y (Map 4, Annex 1; Table 4, Annex 2). Pastures in this category are predominantly found in tropical humid and subhumid climates, as well as in parts of Western Europe and the United States. Because biomass production is steady or seasonal, such pastures are predominantly fenced in and grazed throughout the year.

Intensive cultivated pasture production is found where climatic, economic and institutional conditions are favourable, and land is scarce. Such conditions are typically found in the EU, North America, Japan and the Republic of Korea. In the EU, meat and dairy production units rely to a large extent on temporary pastures (leys), and on the cultivation of forage crops for fresh and conserved feed. The most intensive pastures are found in southern England, Belgium, the Netherlands and parts of France and Germany. Forage systems are high-yield oriented, with regular use of high levels of mineral fertilizers combined with regular manure applications and mechanization. These intensively used pastures are a main source of nutrient loading and nitrate pollution in those countries. Cultivated grasslands are usually species-poor and are typically dominated by *Lolium* species (European Commission, 2004). Intensive forage production in some cases supplies processing industries, such as alfalfa dehydration and hay compaction. These industries (mostly in Canada and the United States) are highly export-oriented.

2.3.2 Feedcrops and crop residues

The feed use of primary food crop products such as cereals and pulses has increased rapidly over recent decades, responding to the growing demand for feed and the inability of traditional feed resources to supply the quantities and qualities required. The growing demand for food and feed has been met without an increase in prices. On the contrary, it was driven by a decrease in cereal prices. In real terms (at constant US$) international prices for grains have halved since 1961 (FAO, 2006b). Expanding supply at declining prices has been brought about mainly by intensification of the existing cropped area.

Cereals

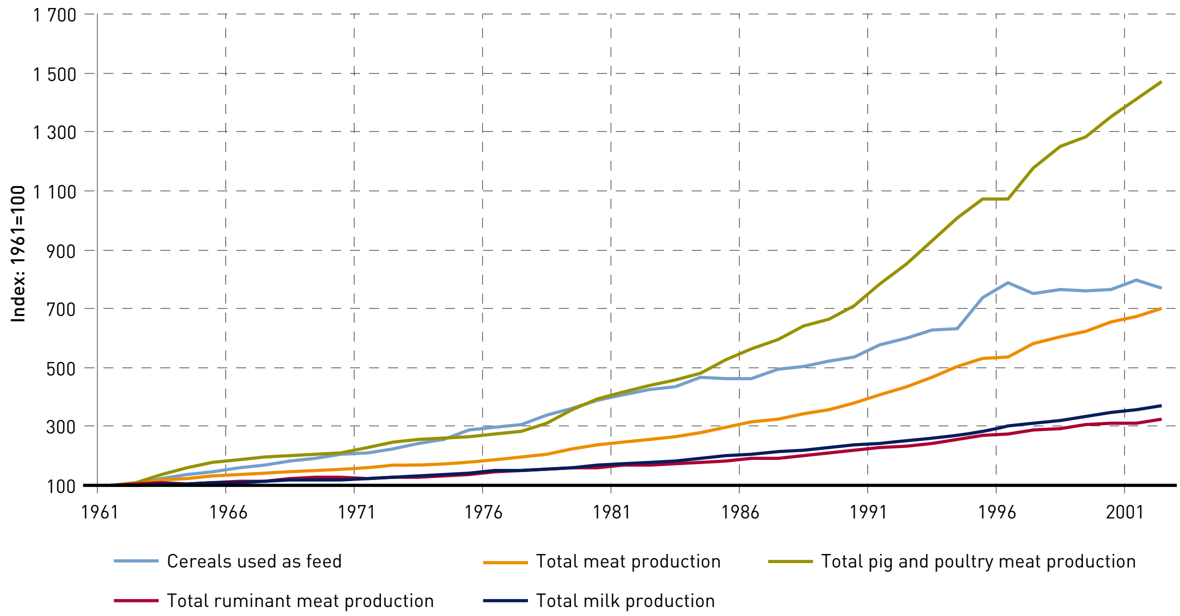
Expansion of feed use slows as feed conversion improves

Some 670 million tonnes of cereals were fed to livestock in 2002, representing a cropped area of around 211 million hectares. A variety of cereals are used as feed, mostly for monogastric species including pigs and poultry. For ruminants, cereals are usually used as a feed supplement. However, in the case of intensive production, such as feedlot or dairy production, they can represent the bulk of the feed basket.

Worldwide, the use of cereal as feed grew faster than total meat production until the mid-1980s. This trend was related to the intensification of the livestock sector in OECD countries, and the related increase in cereal-based animal feeding. During this period, the increasing share of cereals in the feed basket raised the meat production. After this period, meat production has grown faster than cereal use as feed. This can be explained by increasing feed conversion ratios achieved by a shift towards monogastric species, the intensification of livestock production based on high-yielding breeds and improved management practices. In addition, the reduction of subsidies to cereal production under the EU Common Agricultural Policy and economic regression in the ex-socialist countries of Central Europe have reduced the demand for feed grains.

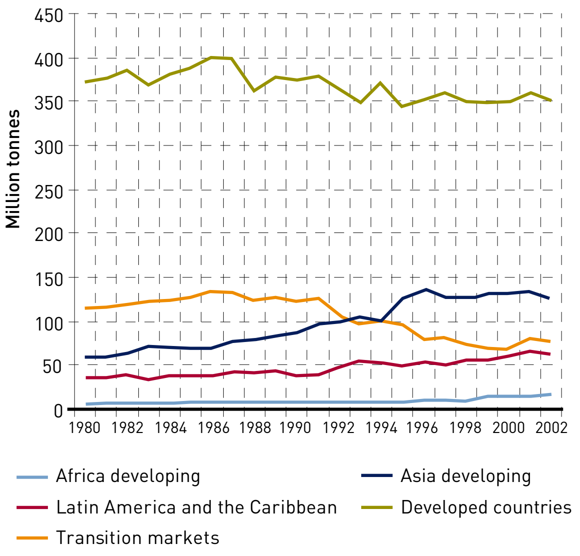
In developing countries, increased meat production has been coupled with increasing use of cereals for feed over the whole period (Figure 2.3). Recently, though, demand for cereal

Figure 2.3 **Comparative growth rates for production of selected animal products and feed grain use in developing countries**



Source: FAO (2006b).

Figure 2.4 **Regional trends in the use of feed grains**



Source: FAO (2006b).

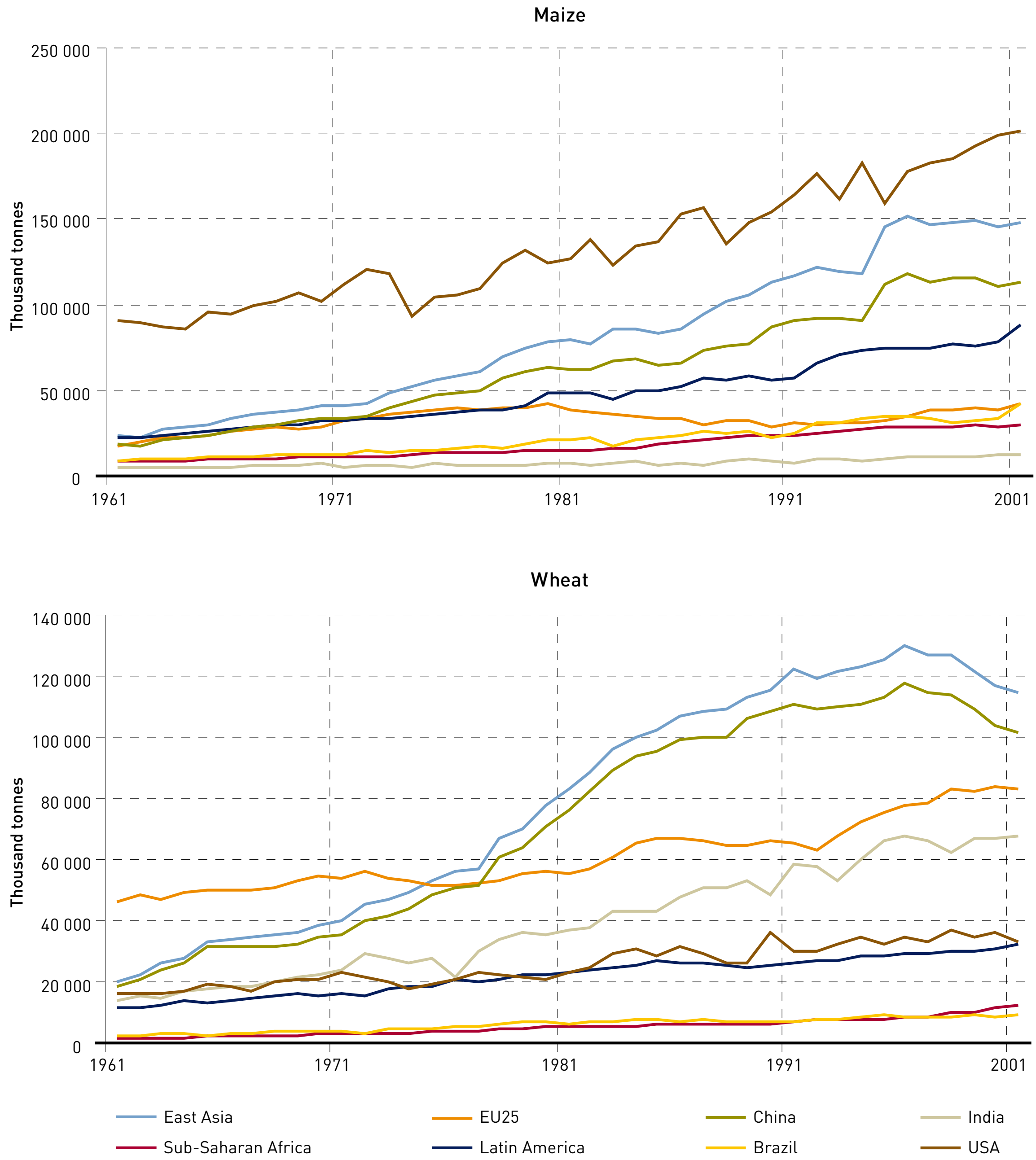
as feed has tended to stabilize, while total meat production has continued to grow, probably driven by highly intensive and monogastric-dominated developing countries, such as Brazil, China and Thailand.

Overall, since the late 1980s, feed demand for cereal has been relatively stable. Such stability, observed at an aggregated level, hides a marked geographical shift in demand, which occurred in the mid-1990s. Demand in the transition countries fell sharply, offset by increases in demand from Asian developing countries (Figure 2.4). At the same time, but more progressively, feed demand dwindled in industrialized countries and strengthened in the developing world.

Expressed as a share of total cereal production, volumes of cereal used as feed increased substantially in the 1960s, but remained fairly stable thereafter and even declined in the late 1990s.

Among the cereals, maize and barley are used mainly as feed – more than 60 percent of their total production over the 1961 to 2001 period. However, feed demand for cereals varies greatly across regions. Maize is the predominant feed cereal in Brazil and the United States, while wheat and barley are dominant in Canada and Europe. Southeast Asia relied on similar proportions of wheat until the early 1990s, since then,

Figure 2.5 **Feed demand for maize and wheat in selected regions and countries from 1961 to 2002**

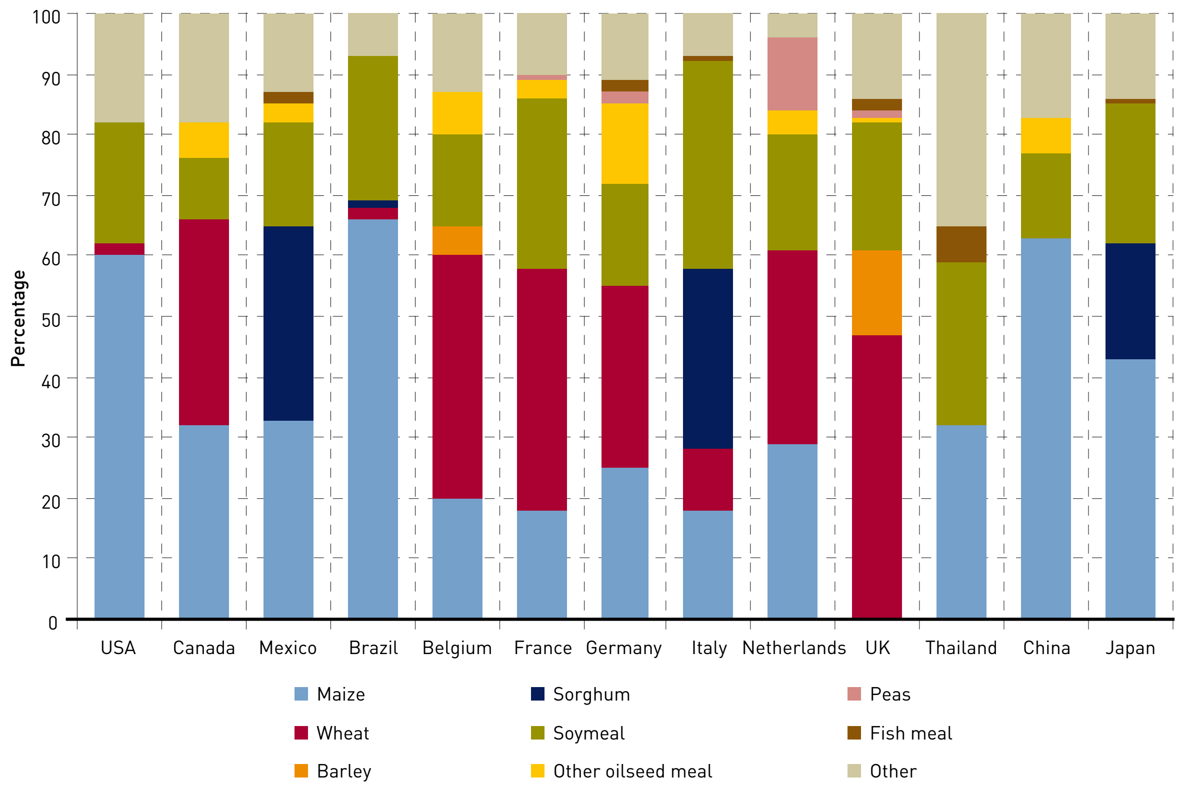


Source: FAO (2006b).

has progressively shifted to maize. These trends reflect the suitability for production of particular crops in these regions – wheat and barley being more adapted to temperate or cold climates than maize (Map 5, Map 6 and Map 7, Annex 1).

Different comparative advantages for producing feedgrains, along with trade conditions, translate into different feed rations at the livestock production level. There is a remarkable homogeneity in the total cereal component in feed rations across analysed countries (cereals represent for example about 60 percent of the weight of chicken feed – Figure 2.6). However, countries differ noticeably in the mix of various

Figure 2.6 **Relative composition of chicken feed ration in selected countries (by weight)**



Note: A large amount of rice is included in the "other" class for Thailand.
Source: Own calculations.

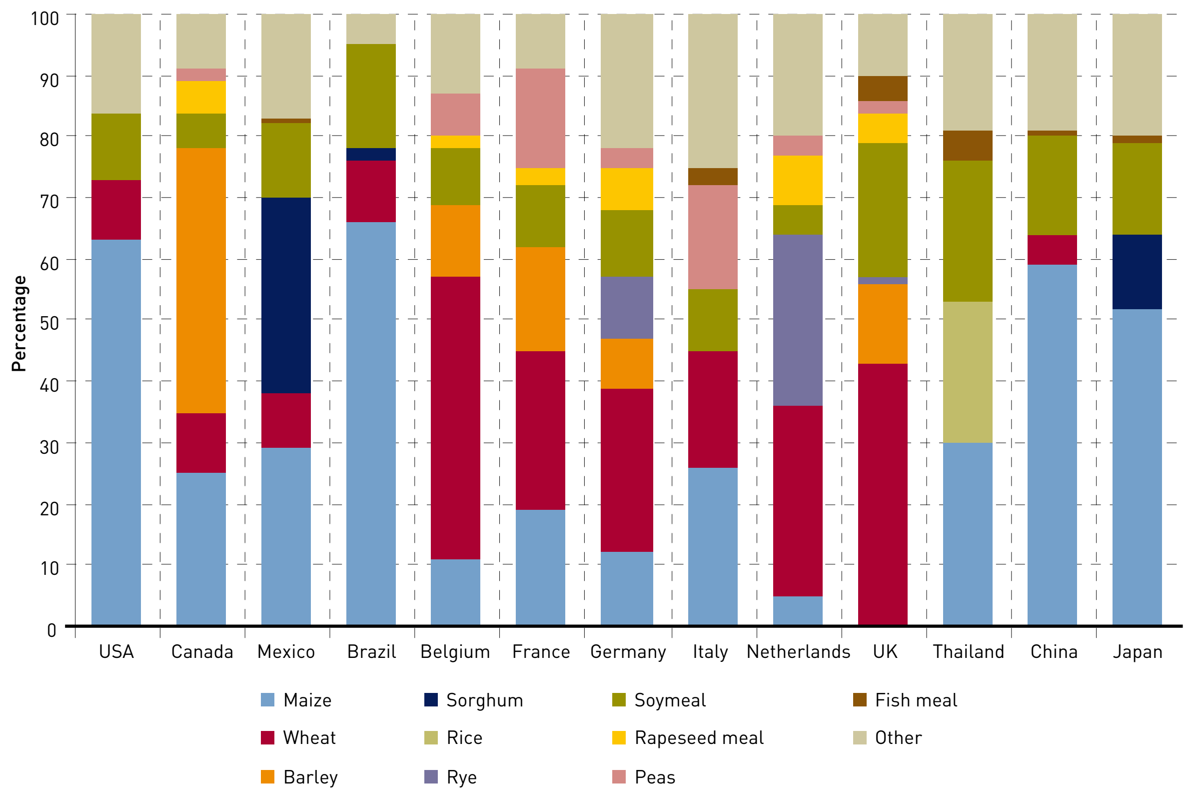
cereals. Maize dominates in chicken feed in Brazil, China and the United States and wheat in the EU. Similar trends are observed for pigs, with a more variable cereal content (60 to 80 percent) for the analysed countries (Figure 2.7).

Crop residues

A valuable but increasingly neglected resource

Crop residues are a by-product of crop agriculture. They are typically high in fibre content but low in other components and indigestibility. The role of crop residues is, therefore, usually one of supplementing basic caloric and fibre requirements, mostly in the diet of ruminants. The use of crop residues such as straw and stover as feed is still fundamental to farming systems that produce both crops and livestock. In these systems, livestock (particularly ruminants) convert residues into valuable food and non-food goods and services. Crop residues represent a large share in the feed basket, especially in tropical semi-arid and subhumid environments where most of the world's poor farmers live (Lenné, Fernandez-Rivera and Bümmel, 2003). Crop residues – as well as agro-industrial by-products – often play a critical role during periods when pastures are in low supply (Rihani, 2005). Devendra and Sevilla (2002) estimated that 672 million tonnes of cereal straws and 67 million tonnes of other crop residues are potentially available as feed in Asia. The actual use of rice straw as feed varies greatly, from over 70 percent of the available total in Bangladesh and Thailand to only 15 percent in South Korea. In other countries of South-east Asia and in China, the share is estimated at between 25 and 30 percent.

Figure 2.7 **Relative composition of pig feed basket in selected countries (by weight)**



Note: A large amount of oats is included in the "other" class for Italy.
Source: Own calculations.

Despite its local importance in smallholder mixed farming systems, the use of crop residues as feed is in decline. A number of factors drive this trend, all are related to agricultural intensification. First, less crop residues are available per unit of crop produced, because of genetic selection aimed at reducing residues (e.g. dwarf cereals) and because of more effective harvesting machinery. Second, genetic selection, based on performance traits relating to the main food product, tends to reduce the quality of crop residues (Lenné, Fernandez-Rivera and Bümmel, 2003). Third, intensive livestock production requires feed of high quality, which typically cannot be provided by crop residues. In addition, crop residues have gained increasing importance as a source of energy and in furniture production.

Other feedcrops

After cereals, the second main category of feedcrop is roots and vegetables. About 45 million tonnes were fed to livestock in 2001 - mostly cassava, potatoes, sweet potatoes, cabbage and plantain. In addition, about 17 million tonnes of pulses (mainly peas and beans) were fed to livestock, representing an important share of protein intake in some places, e.g. France, Italy and the Netherlands. Pulse, root and vegetable feedcrops are estimated to span a total area of over 22 million hectares. Oil seeds can also directly be fed to livestock, although the large majority is processed and only by-products are used as feed. In 2001, feed demand for oil seeds totalled about 14 million tonnes, equivalent to a cropped area of 6.4 million hectares. The main oil seeds used as feed include soybeans, cottonseed, rapeseed and sunflower seed.

2.3.3 Agro-industrial by-products

As humans develop ever more sophisticated food chains, agro-industries are growing and so is the availability of associated by-products as sources of animal feed. An increasing share of human food is being processed, the number of stages of processing is growing, and processing plants are scaling up. All these factors raise the available amounts of by-products of reliable quality, so that gathering and processing them as feed becomes economically profitable.

Soybean

Feed demand drives a production boom

Soymeal, a by-product of the soybean oil industry, is a case in point. In oil extraction, soybeans yield 18 to 19 percent oil and 73 to 74 percent meal (Schnittker, 1997); the rest is waste. Only a small portion of the harvested beans is directly fed to animals (about 3 percent globally). However, more than 97 percent of the soymeal produced globally is fed to livestock. Soymeal is used primarily in the diet of monogastric species, particularly chickens and to a lesser extent pigs. Figure 2.8 shows the high fraction of soybeans processed by the oil industry, and the stable ratio between processed beans and resulting cakes over the last four decades. Worldwide, the feed demand for soymeal has skyrocketed over the past four decades, reaching 130 million tonnes in 2002 – see Figure 2.8. This far outstrips the second largest oilcake, made of rape and mustard seed, with 20.4 million tonnes of production in 2002.

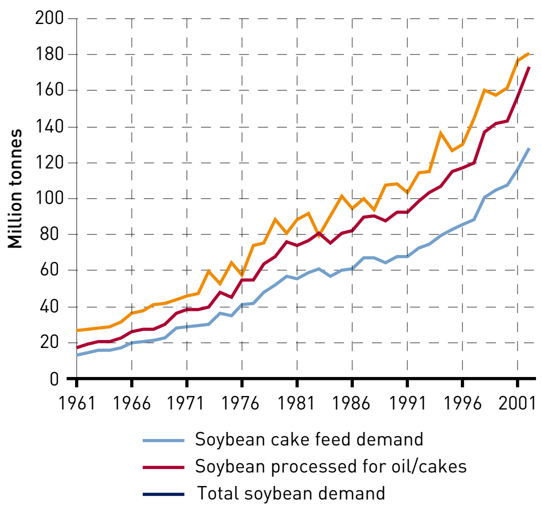
Growth of soymeal feed production took off in the mid-1970s and accelerated in the early 1990s, propelled by rapidly growing demand in developing countries. However, soymeal use per person is much higher in developed countries (50 kg per capita as opposed to 9 kg in developing countries). Over the past four decades, demand for soymeal has increased faster than total meat production, implying a net increase in the use of soymeal per unit of meat produced. This is true for ruminant as well as monogastric species. Part of this increase in use of soymeal by livestock is a consequence of the increasing demand for fishmeal in the fast expanding aquaculture sector, which, with a rather inflexible supply of fishmeal, forced the livestock sector to search for other protein substitutes in livestock feed. Aquaculture is more dependent on fishmeal (and fish oil) than terrestrial animals, and the share of fishmeal used by aquaculture grew from 8 percent in 1988 to about 35 percent in 2000 (Delgado *et al.*, 2003) and 45 percent in 2005 (World Bank, 2005a) despite efforts to reduce the proportion of such products in the fish feed ration. Another factor is the prohibition of using animal offal in animal feed to reduce the risk of mad-cow disease, which put more pressure on the production of vegetable protein for animal feed (see 2.3.4).

World soybean production tripled over the 1984 to 2004 period, half of this increase occurring in the last five years. Production is highly concentrated geographically. Eight countries provide 97 percent of world production; the top three countries (Argentina, Brazil and the United States), account for 39 percent, 26 percent and 17 percent respectively. These three countries also achieved the highest absolute growth in production over the past four decades.

Map 9 (Annex 1) provides an overview of areas where soybean is cropped for oil and meal production. The strong geographical concentration is clearly visible. Soybean processing and marketing have a high level of geographical concentration, specialization, vertical integration and economies of scale. Small producers – especially in developing countries – find it very difficult to compete, especially when faced with the requirements of rapidly expanding and highly efficient international trade. Recently, however, new countries started producing soybeans for export, achieving substantial production growth over the 1999 to 2004 period. These countries are in Latin American (e.g. Bolivia, Ecuador, Uruguay), the former Soviet bloc countries (e.g. Czech Republic, Kyrgyzstan, the Russian Federation and Ukraine) and Africa (e.g. Uganda). Of the largest soybean producers, the United States has the highest average yields: 2.6 tonnes per hectare.

Some of the smaller producers also achieve good results. Argentina and Brazil produce on average about 2.4 tonnes/ha, while China's yields are only 1.65 tonnes/ha. India is far behind with average yields of only 0.90 tonnes/ha (Schnittker, 1997). Over the past decade, the yield increase has been substantial, although most of the extraordinary growth in supply was the result of expansion of soybean harvested area – see Figure 2.2. Although the soy oil industry was initially the main driver of soybean production, feed demand is currently driving the expansion. Indeed, soymeal accounted for about two-thirds of the value of soybeans in recent years, with oil about one-third. This situation has developed over the past 30 to 40 years, as the demand for protein for terrestrial and aquatic animal feed increased rapidly and as the production of other oil-rich seeds such as palm oil, canola and sunflower weakened the demand for soy oil (Schnittker, 1997). This is confirmed by the analysis of feed baskets (Figures 2.6 and 2.7), which shows that soymeal is a major source of protein in all countries analysed. The contribution of other locally produced vegetable protein sources such as peas and other oil cakes is generally limited. The increasing demand for oilseeds for biofuels might change these trends (see 2.3.4).

Figure 2.8 **Global trends in demand for soybean and soybean cake from 1961 to 2002**



Source: FAO (2006b).

Other agro-industrial by-products

Other agro-industrial by-products are less widely commercialized and their use is confined to their regions of origin. They are often used during droughts or other periods of scarce feed supply to supplement pasture and crop residues (Rihani, 2005). In North Africa, their contribution to feed resources for small ruminants rises from 10 percent in favourable years to 23 percent in years with drought, when pasture and crop residues are short (Rihani, 2005). In this region, agro-industrial byproducts used for feed include brewery residues, citrus, tomato and date pulp, olive cakes, and sugarbeet molasses and pulp. In Japan, 30 percent of agro-industrial byproducts are recycled as feed after being dehydrated (Kawashima, 2006).

In contrast, food wastes from marketing and retailing are much less recycled as feed (5 to 9 percent, depending on the source), because their content and quality vary greatly and their geographical spread increases collection costs. The safety of food wastes is also questionable.

Household waste

The use of household waste as feed remains predominant among rural households in developing countries; though in OECD countries it is only sporadic. Food wastes are often collected from food processors in urban centres. Food wastes from individual households have been an important traditional feed resource, in particular for smallholder monogastric and dairy production. Indeed, the recycling of household wastes, as feed for monogastric species, explains the close spatial correlation between human populations and those of pigs and poultry prior to and during

Table 2.8

Supply and recycling of food by-products in Japan

	Supply of by-products per year	Share recycle as feed	Share recycle in other forms
	(thousand tonnes)	*(%)*	*(%)*
Food manufacturing industry	4 870	30	48
Food wholesaler/retailer	3 360	9	26
Food service industry	3 120	5	14
Total	**11 350**	**17**	**32**

Source: Kawashima (2006).

the early stages of industrialization. However, rising environmental and human health requirements usually bring an end to backyard production in urban and peri-urban areas, once rural areas are connected to urban centres adequately enough to provide sufficient and reliable supplies.

2.3.4 Future trends

Increasing feed demand

Today, feed production is estimated to use approximately 30 percent of the emerged land. Statistics on pasture add up to 34.8 million km^2 globally (26 percent of emerged land) while we estimate that about 4.7 million km^2 of cropland are currently dedicated to feed production (4 percent of emerged land or 33 percent of all cropland). The latter does not include crop residues but includes most agro-industrial byproducts (see methodological note in Annex 3). In comparison, the shares of total meat output from grazing, mixed and intensive landless, are estimated at 8 percent, 46 percent and 45 percent respectively (see Section 2.4). The juxtaposition of these figures gives a sense of the strong intensity gradient along which livestock use land.

Livestock production is projected to increase and with it the demand for animal feed. FAO (2003a) estimates that feed demand for grain will increase by nearly one billion tonnes over the 1997/99 to 2030 period (at growth rates of 1.9 percent a year between 1997/99 and 2015, and 1.6 percent per annum thereafter). Most of this growth will be driven by developing countries, where the use of concentrate feeds is projected to grow faster than meat production. Feed use is expected to remain the most dynamic element driving the world cereal economy, accounting for a growing share of aggregate demand. Use of maize as feed is projected to rise from 625 to 964 million tonnes over the 2002 to 2030 period, with most of the growth occurring in developing countries (265 million tonnes), especially in Southeast Asia (133 million tonnes), Latin America (56 million tonnes) and to a lesser extent in sub-Saharan Africa (33 million tonnes). Projected feedcrop growth rates are higher than over the last 15 years. The projected increasing feed demand for cereals is the result of interacting trends.

First, the current recovery of economic decline in transition economies is expected to be sustained, and with it the growing demand for livestock products. Such demand will fuel production and thus feed demand to levels at least equal to those observed in the early 1990s. Feed demand for cereals is also expected to rise in the EU, boosted by decreasing prices induced by the common agricultural policy (CAP) reform process. The reforms proposed in 1992, and implemented in 1994 (Ray MacSharry reform), brought a 30 percent cut in the cereal intervention price, phased in over three years. These were followed by a further reduction in support prices for cereals, which were agreed to in March 1999 in the framework of Agenda 2000. In parallel, factors reducing demand are expected to weaken. Especially, the gain in feed efficiency is expected to dwindle.

In the past decades, the shift towards monogastric species, especially poultry, which has a higher feed conversion ratio than ruminants (typically 2 to 4 versus 7 kg of grain per kilogram of meat) (Rosegrant, Leach and Gerpucio, 1999); further gains in feed efficiency, from advanced feeding methods (multiple-stage feeding) and breeding, have allowed a substantial increase in feed efficiency, which has contributed to the counterbalancing of the soaring demand for feed. However, it is estimated that the shift towards monogastric species will be slower than over the last 20 years (FAO, 2003a) and room for feeding and breeding improvement also seems limited.

The role aquaculture will play in this process is uncertain. Products from fish fed on similar feed as livestock (e.g. tilapia) may be increasingly substituted for livestock products. Because of their substantially better feed conversion ratio than livestock[5] (typically 1.6 to 1.8 for tilapia), aquaculture may play the role poultry played in the past, depressing feed demand for cereals. Although possible, a significant shift to fish products would, however, require both the organization of supply chains and changes in consumers' preference and would thus probably only occur over a long period.

Although at a slower pace, the number of grazing animals will also increase, requiring more fodder. Tilman *et al*. (2001) estimate a net increase of 2 million km^2 of pasture by 2020 and of 5.4 million km^2 by 2050. While recognizing that pasture expansion will probably occur in Latin America and, to a lesser extent, sub-Saharan Africa, the authors of the current study consider that these figures may be overestimated.

The potential and actual production of vegetative feed resources varies substantially across the globe along with different ecological, economic, technical and policy contexts. The question of how feed supply can meet the demand of a burgeoning livestock sector is of relevance beyond its boundaries. Some aspects of this question are assessed below.

[5] Fish are cold-blooded, use less energy to perform vital functions and do not require the heavy bone structure and energy to move on land. Fish catabolism and reproduction is also more efficient.

Mixed cattle at pasture on a ranch in Obala – Cameroon 1969

Pastures: backs against the wall

Exploring options for pasture expansion, Asner *et al*. (2004) suggest that the expansion of grazing systems into marginal areas has already more or less reached the limits imposed by climate and soil factors. Any significant increase of grassland could, therefore, only take place in areas with high agro-ecological potential.

To see what land-use changes might result from pasture expansion, the current dominant land use in areas with high suitability for pasture but no current use as pasture are identified (see Map 10, Annex 1). Globally, forestry is the predominant current use of this land (nearly 70 percent) and in most of the continents, especially in sub-Saharan Africa (88 percent) and Latin America (87 percent). Cropland is the leading current use in West Asia and North Africa, Eastern Europe and South Asia. Urbanization is of local relevance only, except in Western Europe, where urban areas occupy 11 percent of the land suitable for pasture.

These results suggest that any significant increase of grassland into areas with high agro-ecological potential can, therefore, only occur at the expenses of cropland (which is highly improbable) or through the conversion of forests to pasture, as is currently happening in the humid tropics.

In reality, pasture will most probably keep on losing ground to cropland. This trend is already occurring in a number of places, and in particular in Asia and sub-Saharan Africa, fuelled by an increasing demand for grain. Urbanized areas will also encroach into pasture land, especially in areas with booming populations such as sub-Saharan Africa and Latin America. Encroachment by urban and cropland areas is particularly harmful to pasture-based systems, as it usually takes away the most productive land. This compromises the access to biomass during the dry season, when the less productive land cannot sustain the herd. This often results in overgrazing, increased losses during drought and conflicts between pastoralists and agriculturalists.

Pastures are on the increase in Africa and in Latin America where the land colonization process is still ongoing. The pace of pasture expansion into forests will depend mainly on macro- and microlevel policies in concerned areas. In OECD countries, the total pasture area will be stable or declining, as rangelands are converted to cropland, urban areas and natural ecosystems/recreational areas. Since the prospect of expansion on pastureland is limited, the intensification of pasture production on the most suitable land, and loss of marginal pastures, is likely to continue (Asner *et al*., 2004). It is indeed estimated that there is significant scope for increased grassland production, through improved pastures and enhanced management. In the subhumid areas of Africa, and especially in West Africa, Sumberg (2003) suggests that, on fertile soils with good accessibility, crops and livestock will be integrated, while the most remote areas will be progressively marginalized or even abandoned.

Climate change is also likely to alter grassland-based systems. The impact on natural grasslands will be greater than on cropland, where growing conditions can be more easily manipulated (e.g. by irrigation or wind protection). On dry lands, the impact is projected to be dramatic. Results from a case study in Mali by Butt *et al*. (2004) indicate that climate change could reduce forage yields by as much as 16 to 25 percent by 2030, while crop yields would be less affected, with a maximum of 9 to 17 percent reduction for sorghum. In contrast, pastures located in cold areas are expected to benefit from rising temperatures (FAO, 2006c). An opportunity for pasture expansion exists in transition countries, where extensive areas of abandoned grassland would be available for re-colonization at relatively limited environmental cost.

Croplands

Prospects for yield and land expansion jeopardized by degradation and climate change

Producing more feed will require increasing productivity, increasing production area, or a combination of both. There is a wide consensus that the potential to further raise the yield frontier in cereals and oilseeds is generally large; although yields may have peaked in some areas (e.g. the Ganges basin) (Pingali and Heisey, 1999; FAO, 2003a). In the case of major cereals, the yield frontier of maize would be easiest to shift, through technology transfer from industrialized nations. Pingali and Heisey (1999) estimate that this transfer is most likely to occur in China or other parts of Asia, where rapidly expanding demand for feed maize will make the crop increasingly profitable and where the private sector should be able to make the necessary investments. In contrast, growth in soybean yields may be slower (Purdue University, 2006). There is also remaining potential for expansion of cropland. Currently, arable land plus land under permanent crops is estimated to represent slightly over one-third of the land that is suitable for crop production (FAO, 2003a). It is, therefore, estimated that land expansion will continue to contribute to the growth of primary agricultural output.

The prospects vary considerably by region. The possibility of expanding cropland under grains and soybean is limited in South and Southeast Asia (Pingali and Heisey, 1999). It is more promising in most other continents, especially in Africa and Latin America. The contribution of arable land expansion to crop production over the 1997/99 to 2030 period is projected to be 33 percent in Latin America and the Caribbean, 27 percent in sub-Saharan Africa, 6 percent in South Asia and 5 percent in East Asia (FAO, 2003a). These figures reflect the extent of areas with high potential for cereal production (Map 11, Annex 1), and soybean production (Map 12, Annex 1).

Two major issues jeopardize this overall positive picture. First is the land degradation associated with intensifying and expanding crop production, and its consequences in terms of ecological damage and decreased productivity. Declining productivity trends observed lately in South Asia can be directly linked to the ecological consequences of intensive cropping, including the build-up of salinity, waterlogging, declining soil fertility, increased soil toxicity and increased pest populations (Pingali and Heisey, 1999). Expanding arable land into natural ecosystems also has dramatic ecological implications, including loss of biodiversity and of ecosystem services such as water regulation and erosion control. Issues of land degradation associated with intensive agriculture are further investigated in Section 2.5 below.

Second, although there seems to be enough production potential for the world taken as a whole, there are considerable local variations. Because of land scarcity and poor land suitability for cropping, local level land shortages are likely to arise (FAO, 2003a). The impact of climate change will also vary considerably by region. Climate change will affect the yields of vegetative resources for livestock production, mainly through changes in temperature, rainfall, CO_2 concentration, ultraviolet radiation and pest distribution. Indirect effects may also occur through the alteration of soil biology and chemistry. Some of these changes will be damaging, such as reduced yields in many areas; some may be beneficial, such as the "fertilizing effect" of increased CO_2 concentrations. The literature tends to agree that there may be a net reduction of yields aggregated at global level. However, North America, South America, Western Europe and Oceania are often listed among the regions for which climate change may bring increasing yields (Parry *et al.*, 2004).

Competitions and complementarities in the quest for feed biomass

Animals are not the sole users of crops, crop wastes and by-products. The foodcrop, aquaculture, forestry and energy sectors are competing users, thus indirectly competing with livestock for land resources. Direct competition between feed and food demand for cereal is estimated to be low on average. The elasticity of the livestock demand for cereals and oilseeds is much higher than elasticity of the human demand. Thus, when crop prices rise, the demand for meat, milk and eggs tends to decrease rapidly, releasing more of the cereal supply to human consumption. It can, therefore, be argued that the use of cereals by livestock represents a buffer, acting to protect food demand from fluctuations in production (Speedy, 2003). This buffering effect occurs also on a smaller scale, for example with sheep fattening in the Sahel. In a good year, the surplus grain crop is used for the household fattening of sheep, whereas in a bad year, it is exclusively used for human food. But the availability of using grain for animal feed in good years induces farmers to grow more than strictly needed, thus improving food security in a poor year.

FAO projections suggest that, despite regionally contrasted trends, the share of cereals globally used as feed is likely to increase by 2030, driving cereal production growth from 1.8 to 2.6 billion tonnes between 1999/01 and 2030. An increasing share of this feed use will be taken by the aquaculture industry, which is expected to grow at 4 to 6 percent per year to 2015, and 2 to 4 percent per year over the following 15 years (FAO, 1997).

Indeed, with feed conversion ratios better than those for livestock, aquaculture will become a significant competitor to monogastric species in regions such as Southeast Asia and sub-Saharan Africa.

The energy sector is another competitor. With the approaching depletion of fossil fuel resources and increasing efforts to mitigate climate change, green energies based on vegetal biomass are taking off. Today, ethanol produced from sugar cane accounts for 40 percent of the fuel sold in Brazil. Worldwide, fuel ethanol production increased from 20 billion litres in 2000 to 40 billion litres in 2005, and is expected to reach 65 billion litres in 2010 (Berg, 2004,) In 2005, the total area used for biofuel crop production in the EU was around 1.8 million hectares (EU, 2006). The average ethanol yield ranges between 3 000 litres/ha (based on maize) and 7 000 litres/ha (beet) (Berg, 2004). In the medium to long term, this land use may well compete with feed production. It is, however, foreseen that the "second generation" of bio-fuels will rely on a different biomass resource, shifting to the fermentation of lingo-cellulosic materials. If such prospects materialize, the biofuel sector may well become a strong competitor of the grass-based livestock production for the access to biomass.

Complementarities also exist. The potential complementarities between food and feed production at the level of crop residues and agro-industrial by-products are well known and to some extent achieved (e.g. oilseed meal). The further expansion of agro-industrial by-products and non-conventional feed resources may represent a major potential for increasing feed resources from primary crop production.

In contrast, food wastes are seldom recycled as feed. With a very low self-sufficiency for feed (24 percent), Japan is exploring ways of increasing recycling of food waste for feed. In addition to reducing feedstuff imports, the aim is to reduce environmental impacts currently associated with incineration or dumping in landfills. Kawashima (2006) proposes technical options for the sanitation and homogenization of food wastes, based on dehydration, heat treatment and silage.

In various contexts, food wastes and agro industrial by-products could contribute substantially to the feed supply, and by the same token release pressure on land. Their better recycling can help to improve self-sufficiency for feed and to improve animal productivity by supplementing diets. There is also an ecological interest in recycling the nutrients and energy embodied in food wastes and by-products, instead of disposing of them in environmentally damaging ways. However, food safety and ethical concerns do limit the potential for this practice, and must be adequately addressed.

Food safety and consumer preferences also shift feed requirements

The bovine spongiform encephalopathy (BSE) scare has shown the dramatic consequences of an ill-considered recycling of agro-industrial by-products (in this case meat and bone meal) as animal feed. The incident and its media coverage have also brought new livestock feeding practices to general public attention. This and similar events such as dioxin contamination of broiler meat in some EU countries have created widespread consumer distrust in the industrial livestock sector. Following the precautionary principle (UN, 1992), the EU set a ban on feeding meat-and-bone meal to all farm animals starting on 1 January 2001.

While the adoption of the precautionary principle should guarantee safer animal-derived foods, it may have a significant impact on feed requirements. The EU meat-and-bone meal ban is a dramatic example. Before the ban the amount of meat and bone meal consumed in the EU was about 2.5 million tonnes annually. Based on protein equivalency, this equates to 2.9 million tonnes of soymeal or to 3.7 million tonnes of soybeans (USDA/FAS, 2000). Largely because of the ban, EU soymeal imports increased by almost 3 million tonnes between 2001 and 2003, about 50 percent more than over the previous period of the same length. Soybean expansion and shipment creates environmental impacts in terms of biodiversity erosion, pollution and greenhouse gas emissions (see Chapter 3). Although soymeal is the main beneficiary of the meat-and-bone meal ban, corn gluten, field peas, rapeseed meal and sunflower seed meal are other potential substitutes. This example casts a dramatic light on the conflicting objectives associated with livestock production.

The need to address such tradeoffs is likely to become increasingly acute, and policy decisions in this area will be critical to the environmental and social sustainability of the sector. Another factor affecting the feed sector, and in particular the soybean market is consumer concern about genetically modified organisms (GMOs). Responding to consumer concerns, the EU has required that products containing GMOs be labelled so that consumers can identify them. In addition, the EU is pushing for GMO soybeans to be separated from other varieties, so that those purchasing them for feed or as ingredients can make a choice. This trend, if maintained, will impact producers' relative competitiveness as well as production practices. More generally, the use or banning of GMOs in animal feeds will have an impact on the crop species used, production practices, competitiveness of smallholders, yields and the future geographical distribution of their production areas.

2.4 Production systems: location economics at play

Production and processing systems are shaped by the requirements of linking demand with resources (feed, labour, water, etc.), given the available technology and capital. This has resulted in the diverse geographical trends of livestock and production systems that we currently observe. The pattern has changed over time, following human population dynamics (e.g. growth, movements), technical changes (e.g. domestication, cropping, transport) and cultural preferences.

These geographical shifts are still continuing, perhaps even accelerating, as a result of the rapid evolution driven by demand, resource scarcity, technology and global trade (see Chapter 1). The major changes in demand for animal products were reviewed in Section 2.2. They have resulted in a geographical redistribution of demand, with urban centres in rapidly growing economies emerging as consumption centres.

Resource availability influences livestock production costs, especially land and water resources. Previous sections have shown that in several regions of the world there is increasing competition for land and limited options for expanding the feed base, while in other regions there is still potential for expansion. In this section, we will first review the current geographical distribution of livestock and their production systems, in the light of the sector's history. We will then explore current spatial trends of landless and land-based production systems.

2.4.1 Historical trends and distribution patterns

Historically, transport and communication infrastructures were more limited than today. Products were not easily transported and technologies were not propagated rapidly. As a result, demand and resources had to be linked locally, mostly relying on locally available capital and technology mixes. Traditionally, livestock production was based on locally available feed resources, particularly those of limited or no alternative value, such as natural pasture and crop residues. In a context of less developed communication than nowadays, cultures and religions were less widespread and more specific to limited areas. They, therefore, influenced consumer preferences and production options in more diversified ways.

Livestock production systems

Production environments, intensities and goals vary greatly within and across countries. Animal agriculture systems correspond to agro-ecological opportunities and demand for livestock commodities. In general, the systems are adjusted to the prevailing biophysical and socio-cultural environment and, traditionally, since there were no external inputs they have been, for the most part, in sustainable equilibrium with such environments.

In many of these systems, the livestock element is interwoven with crop production, as in the rice/buffalo or cereal/cattle systems of Asia. Animal manure is often essential in maintaining soil fertility, and the role of animals in nutrient cycling is often an important motivation for keeping animals, particularly where this involves a transfer of nutrients from common property resources to private land. In other cases, mobile forms of livestock production have been developed to harness resources from semi-arid or mountainous, seasonally shifting or temporarily available pastures. Although many of these systems result from a long historical evolution, they are currently under pressure to adjust to rapidly evolving socio-economic conditions. Over recent decades, large intensive livestock production units, in particular for pig and poultry production have emerged in many developing regions in response to rapidly growing demand for livestock products.

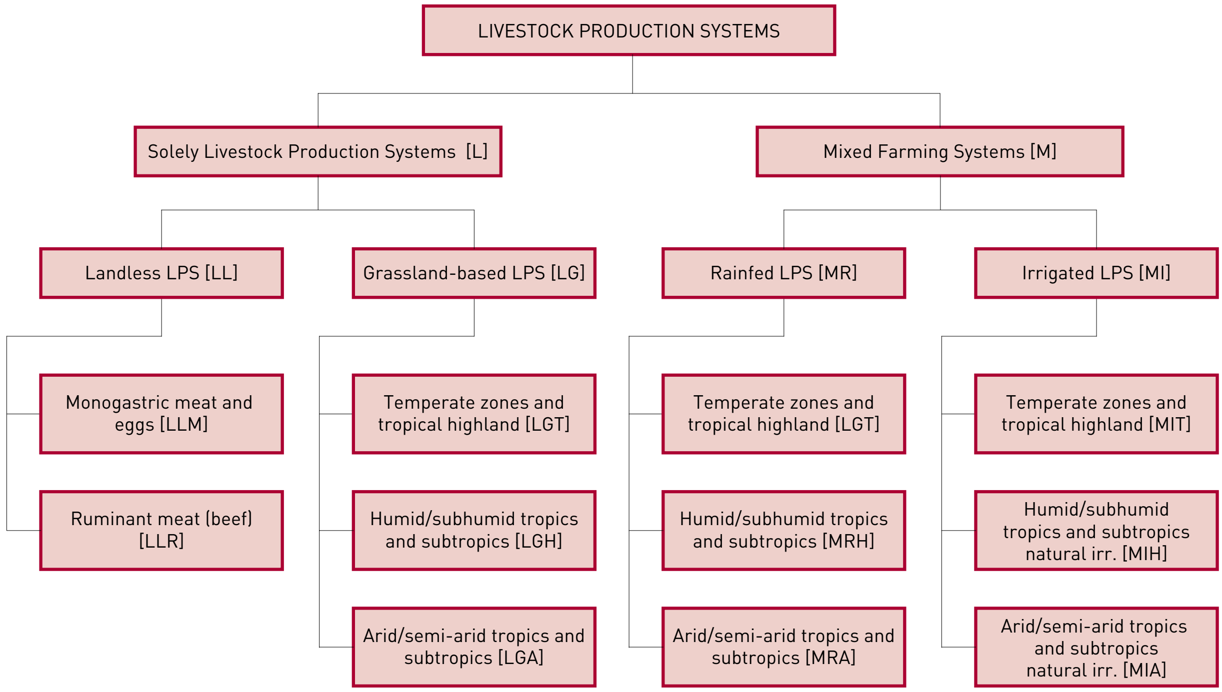
For clarity of analysis, it helps to classify the vast variety of individual situations into a limited number of distinct livestock production systems. Ideally the following criteria should be considered:

- degree of integration with crops;
- relation to land;
- agro-ecological zone;
- intensity of production;
- irrigation or rainfed; and
- type of product.

FAO (1996) has proposed a classification of eleven categories of livestock production systems (LPSs) based on different types of farming systems, relationship to land and agro-ecological zone (see Figure 2.9). They identify two main groups of LPSs:

- those solely based on animal production, where more than 90 percent of dry matter fed to animals comes from rangelands, pastures, annual forages and purchased feeds, and less than 10 percent of the total value of production comes from non-livestock farming activities; and
- those where cropping and livestock rearing are associated in mixed farming systems, where more than 10 percent of the dry matter fed to animals comes from crop by-products or stubble, or more than 10 percent of the total value of production comes from non-livestock farming activities.

Figure 2.9 **Classification of livestock production systems**



Source: FAO (1996).

Below the division between livestock-only and mixed farming, four broad groupings can be distinguished. Map 13 (Annex 1) shows the relative predominance of these four broad groups of livestock production systems around the world (Steinfeld, Wassenaar and Jutzi, 2006), while Tables 2.9 and 2.10 show their relative prevalence in livestock numbers and production data. Two of these broad groupings are among the livestock-only systems: landless LPSs, and grassland-based LPSs.

Landless LPSs are mostly intensive systems that buy in their feed from other enterprises. They are found mostly in Eastern North America, Europe, Southeast and East Asia. These are defined as systems in which less than 10 percent of the dry matter fed to animals is farm-produced, and in which annual average stocking rates are above ten livestock units per km^2 (on average at the census unit level). The landless category defined by FAO (1996) is split into landless ruminant and landless monogastric systems. The presence of landless or "industrial" LPSs is connected to both demand factors and supply determinants. They are prevalent in areas with high population density and purchasing power, in particular coastal areas in East Asia, Europe and North America, that are also connected to ocean ports for feed imports. In contrast, there are areas with ample feed supply such as the mid-western United States and interior parts of Argentina and Brazil, where industrial systems have been developed primarily using local surpluses of feed supplies. East and Southeast Asia strongly dominate industrial monogastric production in the developing regions. Southern Brazil is another industrial production hotspot of global importance. Regionally important centres of industrial production are found, for example, in Chile, Colombia, Mexico and Venezuela, as well as for chicken in the Near East, Nigeria and South Africa.

The other three major categories are land-based, with each category split into three

Table 2.9

Global livestock population and production in different production systems

Parameter	Livestock production system			
	Grazing	Rainfed mixed	Irrigated mixed	Landless/ industrial
Population (million head)				
Cattle and buffaloes	406.0	641.0	450.0	29.0
Sheep and goats	590.0	632.0	546.0	9.0
Production (million tonnes)				
Beef	14.6	29.3	12.9	3.9
Mutton	3.8	4.0	4.0	0.1
Pork	0.8	12.5	29.1	52.8
Poultry meat	1.2	8.0	11.7	52.8
Milk	71.5	319.2	203.7	–
Eggs	0.5	5.6	17.1	35.7

Note: Global averages 2001 to 2003.
Source: Own calculations.

depending on the agro-ecological zone: temperate and tropical highland; humid/subhumid tropics and subtropics; and arid/semi-arid tropics and subtropics

Grassland-based (or grazing) systems are livestock-only LPSs, often based on grazing of animals on seasonal, shifting or upland pastures, primarily found in the more marginal areas that are unfit for cropping because of low temperature, low rainfall or topography, and predominant in semi-arid and arid areas. They are defined as systems in which more than 10 percent of the dry matter fed to animals is farm-produced and in which annual average stocking rates are less than ten livestock units per hectare of agricultural land. These systems cover the largest land area and are currently estimated to occupy some 26 percent of the earth's ice-free land surface. This figure includes a large variety of agro-ecological contexts with very different levels of biomass productivity.

The other two types of land-based system practise mixed crop and livestock farming. Mixed systems are prevalent in bio-climatically more favoured ecosystems.

Rainfed mixed farming systems are mixed systems in which more than 90 percent of the value of non-livestock farm production comes from rainfed land use. Most mixed farming systems are rain-fed and are particularly present in semi-arid and subhumid areas of the tropics and in temperate zones.

Irrigated mixed farming systems are found throughout the world, but have generally limited spatial extent. Exceptions are eastern China, northern India and Pakistan, where mixed irrigated systems extend over large areas. They are defined as mixed systems in which more than 10 percent of the value of non-livestock farm production comes from irrigated land use.

Tables 2.9 and 2.10 show the distribution of production (ruminants and monogastrics) and of animal numbers (ruminants only) over the production system groups, both globally and for the developing regions. The 1.5 billion head of cattle and buffaloes, and the 1.7 billion sheep and goats, are fairly evenly distributed across the land-based systems. However, their average densities increase steeply from grazing systems to mixed irrigated systems, since the latter have far greater livestock-supporting capacities per unit area.

Table 2.10

Livestock population and production in different production systems in developing countries

Parameter	Livestock production system			
	Grazing	Rainfed mixed	Irrigated mixed	Landless/ industrial
Population (million head)				
Cattle and buffaloes	342.0	444.0	416.0	1.0
Sheep and goat	405.0	500.0	474.0	9.0
Production (million tonnes)				
Beef	9.8	11.5	9.4	0.2
Mutton	2.3	2.7	3.4	0.1
Pork	0.6	3.2	26.6	26.6
Poultry meat	0.8	3.6	9.7	25.2
Milk	43.8	69.2	130.8	0.0
Eggs	0.4	2.4	15.6	21.6

Source: Own calculations.

Monogastrics shift towards landless industrial systems, ruminants remain land-based

As yet, only a small fraction of the world's **ruminant** population is found in industrial feedlots, partly owing to the fact that even in intensive production environments feedlots are usually used only in the final stage of the animal's life cycle. The vast majority of large and small ruminant populations are found in the developing regions. Ruminant productivity varies considerably within each system, but overall productivity in developing countries' grazing and mixed systems is lower than in developed countries: globally, beef production per animal in grazing systems is 36 kg/head and year while the average for developing countries is 29 kg/head and year. By far the largest variation in intensity of production is found within the mixed rainfed system, the largest producer of ruminant products. Despite the fact that the developing regions house the vast majority of animals in this category, they account for less than half of the category's production globally. In fact, beef productivity in these regions averages 26 kg/head, as opposed to 46 kg/head at world level, and their milk production is only 22 percent of the world total. Across all four categories, developing regions account for half of the world's beef production, some 70 percent of mutton production and about 40 percent of milk production.

A sharply contrasting situation is found in the **monogastrics** sector. Currently more than half of the world's pork production originates from industrial systems and for poultry meat this share amounts to over 70 percent. About half of the industrial production originates from developing countries and, though reliable population figures are not available, variation in productivity between regions is probably much lower than for ruminants. However, huge differences in total production are found between the developing regions. The majority of the world's pork, poultry and egg production from irrigated mixed systems takes place in developing regions. Although substantial, production in Latin America is less than one-tenth of that in Asia, whereas production is almost absent in Africa and West Asia. The developed countries and Asia together account for over 95 percent of the world's industrial pork production .

Geographical distribution of main livestock species

The distribution of species can also be examined by agro-ecological zone (Table 2.11). Recent strong industrial growth in production of mono-

Table 2.11

Livestock population and production in different agro-ecological zones

Parameter	Agro-ecological zones		
	Arid and semi-arid tropics and sub-tropics	Humid and sub-humid tropics and sub-tropics	Temperate and tropical highlands
Population (million head)			
Cattle and buffaloes	515	603	381
Sheep and goat	810	405	552
Production (million tonnes)			
Beef	11.7	18.1	27.1
Mutton	4.5	2.3	5.1
Pork	4.7	19.4	18.4
Poultry meat	4.2	8.1	8.6
Milk	177.2	73.6	343.5
Eggs	4.65	10.2	8.3

Note: Global averages 2001 to 2003.
Source: Own calculations.

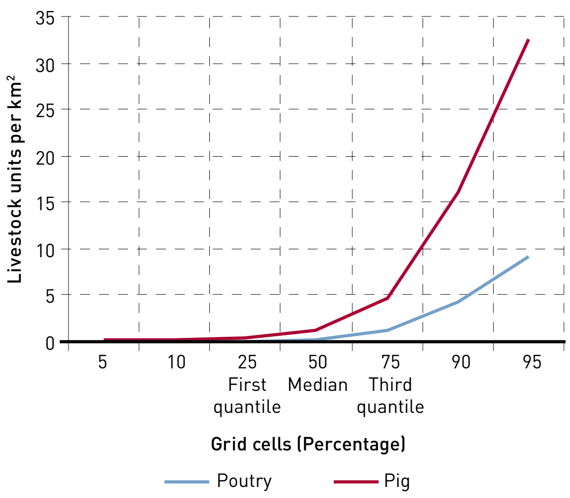
gastrics in the tropics and subtropics has led to production levels that are similar to that of temperate regions. The situation is very different for ruminant production, partly because of its land-based nature; production and productivity are much higher in the cooler climates. Small ruminant production in the (semi)arid (sub)tropics is a notable exception, explained by the large population and the relatively high productivity, the latter being the result of the species' fitness under harsh and marginal conditions. The relatively low productivity for milk in the more humid tropics relates to the strong dominance of mixed systems in these regions, where use of animals for draught power and other uses such as transport is still substantial.

Of all livestock species, poultry has the closest distribution pattern to human populations (see Map 16, Annex 1). This may seem surprising, as poultry is predominantly raised in intensive systems, but the reason is that intensive systems are widely spread. On a global average, three birds are found per hectare of agriculture land, with the highest concentrations found in Western Europe (7.5 birds/ha), East and Southeast Asia (4.4) and North America (4.3). China counts 6.9 birds per hectare of agriculture land. When related to human population, the highest poultry/person ratios are found in North America (6.7 birds per person), followed by Latin America at only 4.5 birds per person. This is consistent with high poultry exports from these two regions (see Table 14, Annex 2).

Historically, the distribution of pig populations was closely related to that of humans. The high concentration of the pig industry in specialized regions has lead to strong subnational concentrations (see Map 17, Annex 1). The tendency for pigs to be more concentrated than poultry in areas with high animal densities is also illustrated in Figure 2.10. This trend may result from the high environmental impact of pig production. The other striking feature of pig distribution is their relative absence from three regions (West Asia and North Africa, sub-Saharan Africa and South Asia) for cultural reasons – see Table 7, Annex 2. On the other hand, the highest pig densities in relation to agricultural land and human population are recorded in Europe and Southeast Asia.

Major **cattle** densities are found in India (with an average of more than one head of cattle per

Figure 2.10 **Comparative distribution of pig and poultry**



Source: Calculation based on Maps 16 and 17 (Annex 1).

hectare of agriculture land), northeastern China (mostly dairy), Northern Europe, Southern Brazil and the East African Highlands (see Map 18, Annex 1 and Table 8, Annex 2). Smaller concentrations are also found in the United States, Central America and Southern China. Although large concentrations are not recorded in Oceania, the region has more cattle than inhabitants, especially in Australia where the cattle population is about 50 percent greater than the human population. Average stock per agricultural land here is, however, among the lowest, in line with the extensive nature of cattle production.

Small ruminants are uncommon in the Americas, except for Uruguay and, to a lesser extent, Mexico and Northern Brazil (see Map 19, Annex 1 and Table 9, Annex 2). In contrast, high densities are found in South Asia and Western Europe (1.3 and 0.8 head per hectare of agricultural land respectively), and there are local concentrations in Australia, China, Northern Africa and African dry lands. As in the case of cattle, sub-Saharan Africa shows higher animal to human population ratios than the world average, which is explained by the heavy reliance on ruminants and the low productivity of animals.

Map 20 (Annex 1) shows global geographical trends of aggregated livestock distribution, expressed in terms of livestock units. We observe six major areas of livestock concentration: Central and Eastern United States, Central America, South Brazil and North Argentina, Western and Central Europe, India and China. Four areas have densely concentrated areas of a lesser extent: Eastern Africa, South Africa, Australia and New Zealand.

Recent distribution trends

Monogastrics expand faster than ruminants

The comparisons between two quantifications of the world livestock productions systems study by FAO, (1996) (averages for 1991–93 and for 2001–03) show that significant changes in resource endowments have brought about changes in the nature and extent of production systems. Cattle stocks are slightly up on the world level (5 percent), with a considerable increase in stock numbers for sub-Saharan Africa, Asia and Latin America. A strong drop in animal numbers (almost 50 percent) occurred in the Eastern European and CIS countries following geopolitical changes and the collapse of the Soviet Union.

World output rose by about 10 percent in the period of observation, with very strong differences at regional level. Cattle meat output almost doubled in Asia. In sub-Saharan Africa it increased by 30 percent, in Latin America by 40 percent, and in West Asia and North Africa by about 20 percent, albeit from a lower absolute level. The strongest cattle output increases occurred in mixed systems in the humid zones. At lower overall production levels (see Table 2.9 and 2.10) total meat production from small ruminants increased by about 10 percent, although the overall stock numbers for small ruminants remained fairly constant for the two reference periods. There were inter-regional shifts in distribution. Stock numbers increased considerably in sub-Saharan Africa and Asia, and strongly declined in Latin America, the OECD, and in particular in Eastern Europe and the CIS. The increases occurred mainly in mixed humid systems. The changes in monogastric animal production are more striking. Total pig meat output (the highest meat output per species in 2002) rose by 30 percent at world level, an increase accounted for almost entirely by Asia. Most regions showed increases in pig meat production, although for Eastern Europe and the CIS there was a drop of about 30 percent. Industrial pig meat production grew at about 3 percent per year. Strong increases also occurred in the humid and temperate mixed irrigated systems.

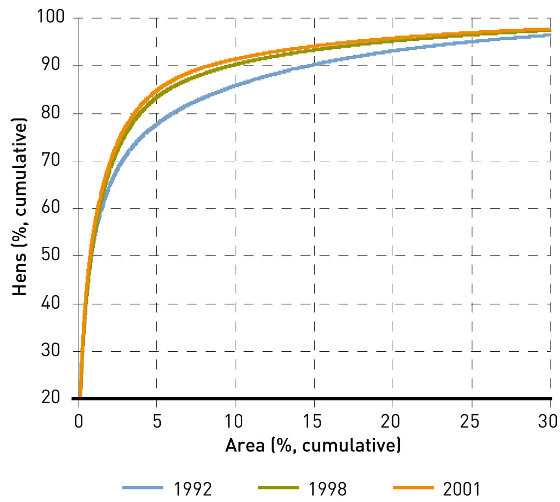
The total production of poultry meat grew by about 75 percent, the strongest expansion of all livestock products. Regional differences were pronounced, with an extremely strong expansion in Asia (about 150 percent increase, with a yearly growth rate of over 9 percent). The growth rates were generally positive, between 2 and 10 percent across regions, most of this resulting from expansion of industrial systems. Global production of table eggs grew by about 40 percent. Asia more than doubled its egg production in the period, to reach a share of about 50 percent of world production. The landless livestock production system grew by about 4 percent per year.

2.4.2 Geographical concentration

The industrialization of livestock production occurs where economic growth is taking place (see Chapter 1). Thus, new farming systems are dominant in industrialized countries and countries with rapid economic growth. A characteristic of such production systems is the segmentation of production stages (feed production, animal raising, slaughtering and processing) and the location of each segment where operating costs are minimized. In this process, animal farms tend to concentrate geographically into clusters.

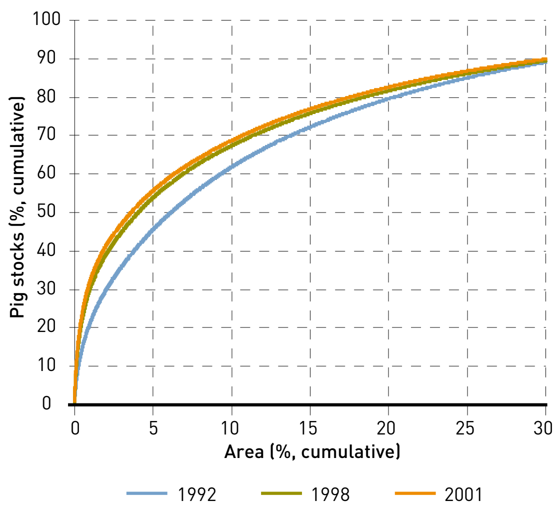
The trend of landless production systems towards clustering is ongoing in developed as well as developing economies. The analysis of the pig and poultry populations at municipal level in Brazil shows a more accentuated geographical concentration for hens than for pigs, and an increasing concentration for both species over the 1992 to 2001 period (see Figures 2.11 and 2.12). In 1992, 5 percent of the total country's area hosted 78 percent of the hen population, rising to 85 percent of the population

Figure 2.11 **Changes in geographical concentration of hens in Brazil from 1992 to 2001**



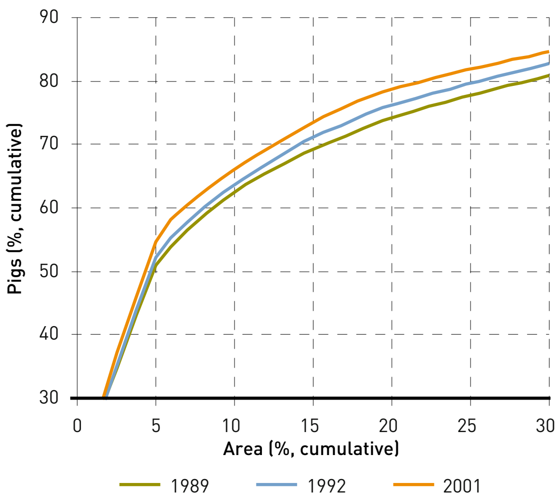
Source: Own calculations.

Figure 2.12 **Changes in geographical concentration of pigs in Brazil from 1992 to 2001**



Source: Own calculations.

Figure 2.13 **Changes in geographical concentration of pigs in France from 1989 to 2001**



Source: Own calculations.

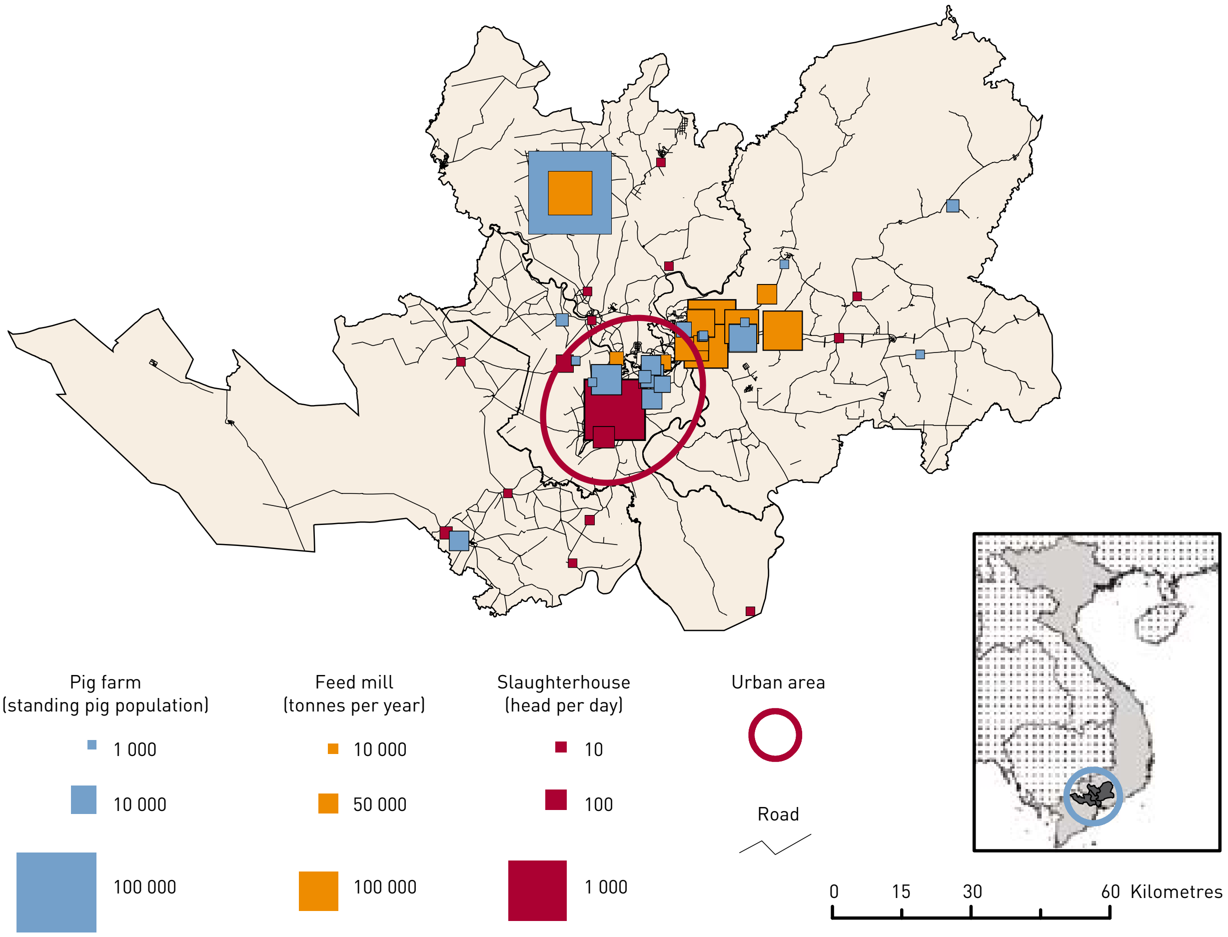
in 2001. The corresponding figures for pigs over the same period are 45 percent and 56 percent respectively. A similar analysis conducted for France and Thailand (see Figures 2.13 and 2.14) showed concurring results.

Landless production systems

A two-step move: rural to urban, urban to sources of feed

As developing countries industrialize, livestock production generally relocates in two stages (Gerber and Steinfeld, 2006). As soon as urbanization and economic growth translate rising population into "bulk" demand for animal food products, large-scale operators emerge. At the initial stage, these are located close to towns and cities. This occurs because livestock products

Map 2.1 **Location of industrial pig sector in southern Viet Nam (Dong Nai, Binh Duong, Ho Chi Minh city and Long An province)**



Source: Tran Thi Dan *et al.*, (2003).

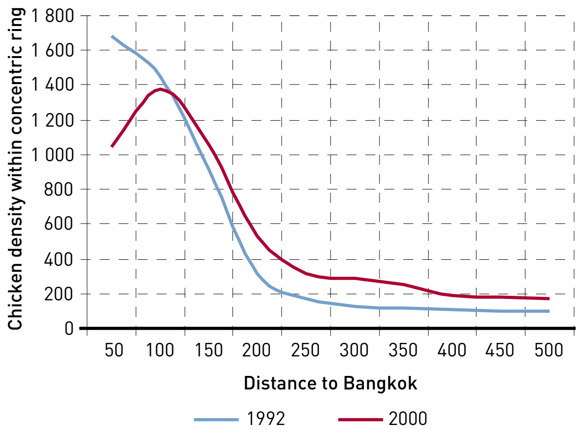
are among the most perishable products, and their conservation/transport without chilling and processing poses serious problems. Therefore, as long as transport infrastructures remain inadequate, livestock-derived foods have to be produced in the vicinity of demand. Map 2.1 illustrates how the intensive pig sector has located at the periphery of Ho Chi Minh City in Viet Nam. Most feed mills, pig farms and slaughterhouses are found within 40 km of the city centre.

In a second phase, transport infrastructure and technology develop sufficiently to make it technically and financially possible to keep livestock further away from demand centres. Livestock production then shifts away from urban areas, driven by a series of factors such as lower land and labour prices, access to feed, lower environmental standards, tax incentives and fewer disease problems. Following a similar trend, the poultry density in areas less than 100 km from Bangkok decreased between 1992 and 2000, with the largest decrease (40 percent) in the areas close to the city (less than 50 km). Poultry density increased in all areas further than 100 km away (see Figure 2.14). In this particular case, the geographical shift was further accelerated by tax incentives.

When pushed out of peri-urban areas, landless production systems tend to move closer to feed resources so as to minimize transport costs on the input side, since the feed used per head is bulkier than the livestock produced. The shift occurs either towards feed production areas (e.g. the United States corn belt, Mato Grosso in Brazil, Mexican El Bajio), or towards feed importing and processing areas (e.g. Chachoengsao Province of Thailand, Jeddah in Saudi Arabia).

In OECD countries, where industrialization of the livestock sector began from 1950 on, clusters formed in rural areas with surplus cereal supply. Here, livestock were initially produced as a means of diversification and value addition. In Europe, pig and poultry production clusters of this type include Brittany, the Po valley in Italy, Western Denmark and Flanders. The geography

Figure 2.14 **Changes in the peri-urban concentration of poultry from 1992 to 2000 in Thailand**



Source: Own calculations.

of these clusters was affected by the increasing use of imported feed. Those with good connection to ports strengthened (e.g. Brittany, western Denmark, Flanders) and new production areas appeared in the vicinity of major ports (Lower Saxony, Netherlands, Catalonia). Finally, a more recent type of feed-related production cluster is observed close to newly created feed processing plants establishing comprehensive animal production chains. Concentration close to feed processing plants is observed in Brazil by analysis of pig numbers and feedcrop production at *Municipio* level in Brazil. From 1992 to 2001, part of the pig population moved away from traditional feed production areas and concentrated around major feed mills in Mato Grosso.

Disease control strategies may, however, scatter production clusters. To limit the spread of diseases, large farms tend to scatter away from other large farms and small-scale units. A distance of a few kilometres is sufficient to prevent disease propagation. It is therefore probable that this trend will prevent the concentration of small- and large-scale farms, especially in peri-urban settings, but will most probably not alter the trend towards specialized areas, equipped with feed mills, slaughterhouses and animal health services.

Land-based systems: towards intensified systems

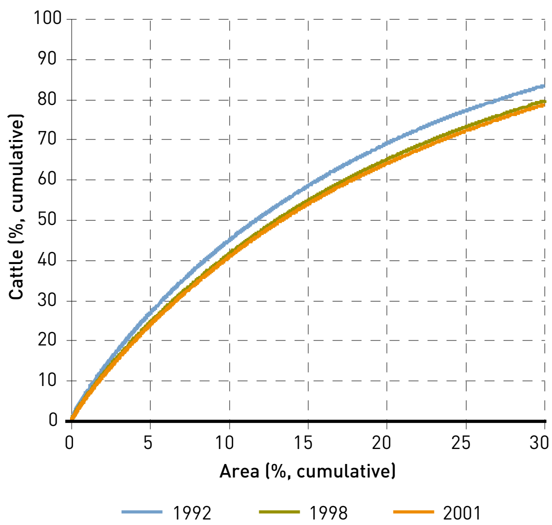
Fodder is bulky and its transportation expensive. Livestock raised in land-based systems are, therefore, bound to feed resource production areas. Previous sections have, however, shown that pasture expansion is likely to be limited, blocked on one side by lack of suitable land and on the other by competition from land uses with lower opportunity costs (e.g. agriculture, forestry, conservation).

As a result, pushed by an increasing demand for beef and milk, part of the production shifts from land based towards intensified systems, such as feedlots and dairy plants (see Chapter 1), following the same geographical trend as intensive monogastric production.

Land-based systems also tend to expand into the remaining areas with good potential for pasture or where there are no strong land use competitors. These are predominantly found in Oceania and South America. Over 1983 to 2003, beef and milk production grew by 136 percent and 196 percent respectively in Oceania, and by 163 percent and 184 percent respectively in South America. For comparison, world production as a whole increased by 124 percent for the two products over the same period (FAO, 2006b).

These overall trends are confirmed by local analysis. Cattle numbers per *Municipio* in Brazil in land-based livestock systems show a more even geographical spread of cattle (see Figure 2.15) than was observed for stock in landless, intensive systems. The expansion of pasture into the Amazon is further described in land degradation hotspots (Section 2.5 below).

Figure 2.15 **Changes in geographical concentration of cattle in Brazil from 1992 to 2001**



Source: Own calculations (2005).

Transporting chickens to poultry plant near Magee – United States

2.4.3 Increasing reliance on transport

Trade and transport improvements increase transport of livestock products

The transport of livestock sector commodities has become increasingly economically affordable and technically possible. Technical changes in transport, such as the development of infrastructure, large-scale shipments of primary crop production or consolidation of long-distance cold chains, have played a determinant role in shaping change in the livestock sector.

Developments in transport have made it possible to bridge the geographical gap between urban demand for animal products and the land resources for their production. Increased trade and transport of animal products and feedstuff are also fundamental factors in the industrialization of the livestock sector. Because they operate on a large scale, with considerable volumes of inputs and outputs, landless industrial systems intrinsically rely on transport for supply of inputs (especially feed) and delivery of outputs. Furthermore, the low private costs of transport (which rarely factor in social and environmental costs) have strongly influenced the location economics of the various segments of the livestock commodity chain, from feed production and feed mill, to animal production, slaughtering and processing. Since the transport cost of connecting each segment is limited, other production costs play a greater role in determining location. Such parameters include cost of land, labour, services, health control, tax regimes and strictness of environmental policy. Although to a lesser extent than landless industrial systems, land-based production systems increasingly rely on transport, as they shift closer to available land resources and further away from consumption centres.

Worldwide, most livestock are produced for national consumption. However, animal products are increasingly traded, and a larger share of the global production enters trade nowadays than in the 1980s. The trend was particularly dynamic for poultry meat, where the internationally traded share rose from 6.5 percent in 1981–83 to 13.1 percent in 2001–03. In 2001–03, more than 12 percent of bovine meat, poultry meat, and milk produced worldwide were traded, and 8.2 percent of pig meat. All of these shares were significantly up on the 1981–83 average. Among feeds, trade in soymeal represented a higher share of production (24–25 percent) over the same periods, though showing little increase (see Table 2.12). For feedgrains the traded share of total production has also remained fairly constant. Trade increases were fostered by a number of policy measures and agreements aimed at easing international trade, including regional trade agreements, harmonization of standards and the inclusion of agriculture in the World Trade Organization (WTO) mandate.

Table 2.12

Trade as a share of total production for selected products

Product	1981–1983 average	2001–2003 average
	(....................... %	.......................)
Bovine meat	9.4	13.0
Pig meat	5.2	8.2
Poultry meat	6.5	13.1
Milk equivalent	8.9	12.3
Soymeals[1]	24.3	25.4

[1] Soymeal trade over soybean production.
Source: FAO (2006b).

Feed trade: Americas dominate exports, China and EU dominate imports

As livestock production grows and intensifies, it depends less on locally available feed resources and more on feed concentrates that are traded domestically and internationally. Map 21 and 22 (Annex 1) display estimated spatial trends in feed surplus/deficit for pig and poultry, providing evidence of the sector's high reliance on trade. Feed trade and the related transfers of virtual water, nutrients and energy is a determinant factor of the sector's environmental impacts. Statistics on feedgrains are generally not separate from overall grain trade flows. However, major trends can be inferred from regional level trade flows, as shown in Table 10 of Annex 2 for maize. North and South America are the two regions with significant interregional exports. The maize that they export to Africa is predominantly used for food, while a large share of exports to Asia, EU and America supplies feed demand (Ke, 2004). Asian maize demand, driven by the feed sector, is predominantly supplied by North America, although imports from South America increased dramatically over the period. North America also exported large volumes of maize to South and Central America (2.8 and 9.2 million tonnes respectively (2001 to 2003 average). Both flows have increased strongly over the past 15 years. On the other hand South America dominates the EU market. Contrasted country profiles and strategies explain these trends. Exports from North and South America are driven by countries (e.g. Argentina, Canada, United States) with ample land resources and strong grain export policies. On the other hand, China, which is a major driver of Asian imports, compensates for its land shortage with imports.

The comparison of grain resources and grain requirements at the local level allow estimating domestic trade (see Map 21, Annex 1), although imports from international markets would most probably supply part of the demand in deficit areas.

About one-third of global soybean, soy oil and soymeal production is traded (29.3, 34.4 and 37.4 percent respectively). This proportion is significantly above that recorded for other agricultural commodities. Soymeal and soybeans account for 35 and 50 percent of the total value of soy-based trade, respectively (FAO, 2004a). The widespread consumption of soybeans is supplied by a few major exporting countries to a large number of importing countries (see Table 11 and Table 12, Annex 2 and Map 22, Annex 1). The United States is the largest soybean exporter (29 million tonnes), followed by Brazil (17 million tonnes). Among the top seven producers, China is the only one with decreasing exports over the period (see Table 11, Annex 2). Indeed, over the past 10 to 20 years China has gone from being a soybean exporter to being the world's largest importer of whole soybeans and a large importer of soymeal – with one-third of its soymeal consumption supplied by imports.

Countries import soybeans either raw, or processed into soy oil and/or soymeal, depending on domestic demand, which is also determined by the structure of the local processing industry. The United States exports about 35 percent of its raw soybeans, before processing. In contrast, Argentina and Brazil add value to most of their crop, process about 80 to 85 percent of their soybeans before export (Schnittker, 1997). For soymeal, South America dominates interregional trade, with the EU as first client and Asia as second (18.9 and 6.3 million tonnes respectively in 2002). The United States has a lesser role in soymeal interregional trade. In recent years, a number of importing countries, especially in the EU have shifted from the importation of soymeal to purchases of beans, which reflects efforts to promote processing at the local level. As a result, about six million tonnes of soymeal produced in the EU enter trade, mostly intraregional, but also towards Eastern Europe. There is also international trade in other fodder products, such as processed alfalfa and compressed hay bales. Exporting countries are predominantly Canada and the United States. Japan is by far the largest importer, followed by the Republic of Korea, and Taiwan Province of China.

Animal and derived products trade increases globally

Live animals and animal-derived products are traded in smaller volumes than feed, because of smaller demand volumes and greater private costs of transport per unit. Nevertheless, the growth of trade in animal products is outpacing the growth of feed trade and of animal production. This rapid growth is facilitated by weakening tariff barriers within the context of GATT, and by the preparation of codes and standards to regulate global trade. In parallel, the trend towards increased demand for processed products by households and catering further expanded the transport of animal products.

Trade in poultry meat has overtaken trade in beef over the past 15 years, with volume soaring from about 2 million tonnes in 1987 to 9 million tonnes in 2002, compared to beef's rise from 4.8 to 7.5 million tonnes over the same period. Except for Eastern Europe, all analysed regions became increasingly involved in trade (see Table 14, Annex 2). North America supplies about half of the interregional market (2.8 million tonnes per year on average, between 2001 and 2003), followed by South America (1.7 million tonnes) and the EU (900 000 tonnes). Brazil is the top exporting country. With relatively low feedgrain and labour costs and increasingly larger economies of scale, Brazil's production costs for whole eviscerated chicken are estimated to be the lowest of any major supplier (USDA-FAS, 2004). On the importer side, the picture is more diversified than for beef, with several regions playing important roles. Asia ranks number one, followed by the Baltic states and CIS, the EU, sub-Saharan Africa and Central America. Important and rapidly increasing regional level trade is taking place in Asia and the EU, both regions yielding local competitive advantages.

To assess transport of meat further, we calculated balances between primary production and demand for animal products at the local level. The results for poultry meat are shown on Map 23 (Annex 1). Production is similar to consumption on a majority of grid cells. A balanced situation (set as +/- 100 kg of meat per km^2) is generally found in land-based systems (compare with Map 13, Annex 1). Areas of highly positive balances (surplus) are associated with landless industrial systems (Map 14, Annex 1), whereas negative balances (deficit) usually coincide with high population densities and urban areas. The poultry exporting position of North and South America shows up here as a dominance of red (surplus) pixels in these two regions. The same analysis conducted for pig meat (Map 24, Annex 1) shows a similar coincidence of positive balances with industrial production areas. However, poultry and pig meat differ in the geographical spread of areas with negative and positive balances. Production areas are generally more scattered among consumption areas for poultry than for pigs. The three maps also show important domestic trade.

Beef is predominantly exported from Oceania and South America, taking advantage of their land-based cattle production systems (Table 13, Annex 2). North America is the main market for Oceania (903 thousand tonnes per year on average, between 2001 and 2003), but Asian imports from Oceania have dramatically increased in recent years (686 thousand tonnes per year on average, between 2001 and 2003, a 173 percent increase in 15 years). South American exports go mainly to the EU (390 thousand tonnes per year on average, between 2001 and 2003) and Asia (270 thousand tonnes), both volumes having roughly doubled over the past 15 years. The EU and North America also make large contributions to global bovine meat supply, based on more intensive production systems, especially in the United States. Most of the EU's trade is within the EU region, although the EU also supplied the Baltic states and CIS countries in 2002. North America predominantly supplies Asia, which is by far the biggest beef importer of all ten analysed regions, importing about 1.8 million tonnes of beef per year on average, between 2001 and 2003 (see Table 13, Annex 2). Asian imports, driven by China, are also the most dynamic, with a 114 percent increase over the 1987 to 2002 period. Asia responds to its soaring demand through interregional trade, but also by drawing upon a booming intraregional beef meat market. Interregional trade is also building up in Sub-Saharan Africa. Finally, Table 13 (Annex 2) illustrates the collapse of Eastern Europe over the period, with imports from North America, sub-Saharan Africa and the Baltic States and CIS that are close to zero. The estimated beef balances (Map 25, Annex 1) display the need for both domestic trade and international trade.

2.5 Hotspots of land degradation

As a major land user, the livestock sector has a substantial influence on land degradation mechanisms in a context of increasing pressure on land (see Box 2.3). With regard to land-based systems, two areas pose the most serious problems. There is the ongoing process of degradation of pastures, particularly in the arid and semi-arid environments of Africa and Asia, but also in subhumid zones of Latin America. There is also the issue of pasture expansion, and the conversion of forest land into pastures, particularly in Latin America.

Landless industrial systems are disconnected from the supporting land base. The separation of production from resources often creates pollution and soil degradation problems, both at feed production and animal operation levels. In parallel, feedcrop expansion into natural ecosystems creates land degradation.

In the following sections, we will review four major mechanisms of land degradation related to the livestock sector:

- expansion into natural ecosystems;
- rangeland degradation;
- contamination in peri-urban environments;
- pollution, soil degradation and productivity losses in feedcrop production areas.

We will assess the geographical extent of these problems, as well as their underlying biophysical process. Impacts on the global environment will simply be listed here. Implications on climate change, water depletion and biodiversity erosion will be further developed in later chapters.

2.5.1 Pastures and feedcrops still expanding into natural ecosystems

Crop and pasture expansion into natural ecosystems has contributed to livestock production growth, and will probably do so in the future under the "business as usual" scenario. Whatever the purpose, the destruction of natural habitats to establish agricultural land use means direct and significant biodiversity losses. The Millennium Ecosystem Assessment (MEA) lists land-use change as the leading cause of biodiversity loss (MEA, 2005a). The destruction of vegetative cover also leads to carbon release, fuelling climate change. In addition, deforestation affects water cycles, reducing infiltration and storage and increasing runoff by the removal of canopies and leaf litter, and through the reduced infiltration capacity of the soil as a result of reduced humus content (Ward and Robinson, 2000).

In OECD countries, the decision to plant soybeans or grain does not usually mean clearing natural habitat. Producers merely make a choice

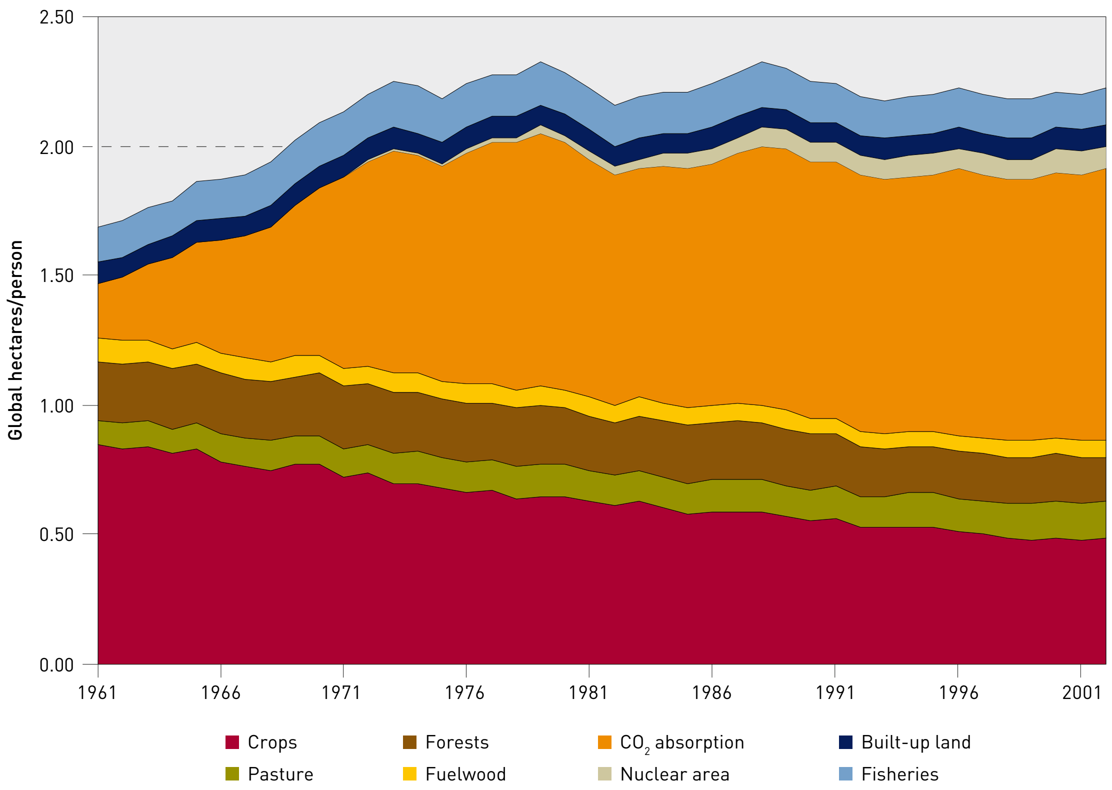
Illegal deforestation for soybean production in Novo Progresso, State of Pará – Brazil 2004

Box 2.3 **Ecological footprint**

To measure humanity's pressure on land and increasing competition for scarce resource, the Global Footprint Network defined an indicator called the ecological footprint. The ecological footprint measures how much land and water area a particular human population requires to produce the resources it consumes and to absorb its wastes, taking into account prevailing technology (Global Footprint Network). This indicator allows us to compare the use of resources with their availability. The Global Footprint Network estimates that global demand for land overtook global supply by the end of the 1980s. It is further estimated that humanity's ecological footprint is currently 20 percent larger than the entire planet can sustain. In other words, it would take one year and two months for the earth to regenerate the resources used by humanity in a single year.

Livestock-related activities contribute significantly to the ecological footprint, directly through land use for pasture and cropping, and also indirectly through the area needed to absorb CO_2 emissions (from fossil fuel use in livestock production) and ocean fisheries (related to fishmeal production for feed).

Figure 2.16 **Ecological footprint per person, by component**



Source: Global Footprint network (available at http://www.footprintnetwork.org).

between a number of crops, within an agricultural area that remains roughly stable. In many tropical countries, however, the cultivation of crops is often driving the process of converting extended areas of natural habitat to agriculture. This is the case in much of tropical Latin America, sub-Saharan Africa and Southeast Asia. Soybeans in particular are a driving force. Between 1994 and 2004, land area devoted to growing soybeans in Latin America more than doubled to 39 million ha, making it the largest area for a single crop, far above maize, which ranks second at 28 million hectares (FAO, 2006b). In 1996, there were only 1 800 hectares of soybeans in Rondônia in the western Amazon, but the area planted increased to 14 000 hectares in 1999. In the eastern Amazon in the state of Maranhão the area planted with soybeans increased from 89 100 to 140 000 hectares between 1996 and 1999 (Fearnside, 2001). The demand for feed, combined with other factors, has triggered increased production and exports of feed from countries like Brazil where land is relatively abundant.

The land area used for extensive grazing in the neotropics has increased continuously over the past decades and most of this has been at the expense of forests. Ranching-induced deforestation is one of the main causes of loss of unique plant and animal species in the tropical rainforests of Central America and South America as well as of carbon release into the atmosphere. Livestock production is projected to be the main land use replacing forest in the neotropics after clearing. Indeed, Wassenaar and colleagues (Wassenaar *et al.*, 2006) estimate that the expansion of pasture into forest is greater than that of cropland. For South America, Map 33B (Annex 1) indicates deforestation hotspots and areas with a more diffuse deforestation pattern. The full ecological and environmental consequences of such deforestation processes are not yet fully understood and deserve greater attention from the scientific community. This is a particularly acute issue, since the major potential for pasture expansion exists predominantly in areas currently under humid and subhumid forest. There is little evidence of the livestock sector being a major factor in deforestation in tropical Africa. Timber harvesting and fire seem to be the two main processes leading to deforestation. Cases of farming replacing forest are predominantly due to small-scale cropping, or to using secondary forest and scrub land for wood harvesting.

Main global environment concerns associated with feedcrop and pasture expansion into natural ecosystems include climate change, through biomass oxidation and carbon release into the atmosphere; water resources depletion through disruption of water cycles and; biodiversity erosion through habitat destruction. These issues will be reviewed in Chapter 3, 4 and 5, respectively.

2.5.2 Rangeland degradation: desertification and vegetation changes

Pasture degradation related to overgrazing by livestock is a frequent and well studied issue. Pasture degradation can potentially take place under all climates and farming systems, and is generally related to a mismatch between livestock density and the capacity of the pasture to be grazed and trampled. Mismanagement is common. Ideally the land/livestock ratio should be continuously adjusted to the conditions of the pasture, especially in dry climates where biomass production is erratic, yet such adjustment is rarely practiced. This is particularly the case in the arid and semi-arid communal grazing areas of the Sahel and Central Asia. In these areas, increasing population and encroachment of arable farming on grazing lands, have severely restricted the mobility and flexibility of the herds, which enabled this adjustment. Pasture degradation results in a series of environment problems, including soil erosion, degradation of vegetation, carbon release from organic matter decomposition, loss of biodiversity owing to habitat changes and impaired water cycles.

Concentrated "hoof action" by livestock – in areas such as stream banks, trails, watering points, salting and feeding sites – causes compaction of wet soils (whether vegetated or exposed) and mechanically disrupts dry and exposed soils. The effects of trampling depend on soil texture – soils with greater fractions of silt and clay are more easily compacted than sandy soils. Compacted and/or impermeable soils can have decreased infiltration rates, and therefore increased volume and velocity of runoff. Soils loosened by livestock during the dry season are a source of sediments at the beginning of the new rainy season. In riparian areas the destabilization of streambanks by livestock activities contributes locally to a high discharge of eroded material. Furthermore, livestock can overgraze vegetation, disrupting its role of trapping and stabilizing soil, and aggravating erosion and pollution. Ruminant species have distinct grazing habits and thus different aptitude to cause overgrazing. For instance, goats being able to graze residual biomass and ligneous species have also the greatest capacity to sap grasslands' resilience. (Mwendera and Mohamed Saleem, 1997; Sundquist, 2003; Redmon, 1999; Engels, 2001; Folliott, 2001; Bellows, 2001; Mosley *et al.*, 1997; Clark Conservation District, 2004).

Asner *et al.* (2004) suggest three types of ecosystem degradation syndromes related to grazing:

- desertification (in arid climates);
- increased woody plant cover in semi-arid, subtropical rangelands; and
- deforestation (in humid climates).

The role of livestock in the deforestation process has been reviewed in Section 2.1 above. Asner and colleagues describe three major elements of desertification: increased bare soil surface area; decreased cover of herbaceous species; and increased cover of woody shrubs and shrub clusters.

The overarching pattern is one of increased spatial heterogeneity of vegetation cover and of soil conditions (e.g. organic matter, nutrients, soil moisture).

Woody encroachment has been well documented in semi-arid, subtropical rangelands of the world. There are hotspots in North and South America, Africa, Australia and elsewhere, where woody vegetation cover has increased significantly during the past few decades. Among the causes are overgrazing of herbaceous species, suppression of fires, atmospheric CO_2 enrichment and nitrogen deposition (Asner *et al.*, 2004; van Auken, 2000; Archer, Schimel and Holland, 1995).

The extent of grassland degradation in arid and semi-arid climates is a serious source of concern and debate, as its quantification is complex. There is a lack of reliable and easily measurable land quality indicators, ecosystems also fluctuate, and the annual vegetation of these arid areas has shown to be highly resilient. For example, after a decade of desertification in the Sahel, there is now evidence of increasing seasonal greenness over large areas for the period 1982 to 2003. While rainfall emerges as the dominant causative factor for the increase in vegetation greenness, there is evidence of another causative factor, hypothetically a human-induced change superimposed on the climate trend. The notion of human induced irreversible degradation of the Sahelian rangelands is thus challenged (Herrmann, Anyamba and Tucker, 2005). On the other hand, desert is rapidly gaining on pasture in northwestern China (Yang *et al.*, 2005). Diverse estimates exist for the extent of desertification. According to the Global Assessment of Human and Induced Soil Degradation methodology, the land area affected by desertification is 1.1 billion ha, which is similar to UNEP estimates (UNEP, 1997). According to UNEP (1991), when rangelands with degraded vegetation are added (2.6 billion ha), the share of dry lands that are degraded is 69.5 percent. According to Oldeman and Van Lynden (1998), the degraded areas for light, moderate and severe degradation are 4.9, 5.0 and 1.4 billion hectares, respectively. How-


© FAO/6077/H. NULL

Soil erosion in the Solo River basin – Indonesia 1971

ever, these studies do not take into account of vegetation degradation. Map 27 (Annex 1) shows the location of grasslands established on weak soils in harsh climates, which face significant risks of degradation if ill-managed.

In addition, there is the risk of pasture degradation in humid to temperate climates. When stocking rates are too high, the removal of nutrients (especially nitrogen and phosphorus) via livestock products and via soil degradation processes may be higher than the inputs, and soils are "mined". In the long run, this results in pasture degradation, evidenced by productivity decline (Bouman, Plant and Nieuwenhuyse, 1999). With decreasing soil fertility, weeds and undesired grass species compete more strongly for light and nutrients. More herbicides and manual labour are needed to control them, which has a negative impact on biodiversity and on farmers' income (Myers and Robbins, 1991). Pasture degradation is a widespread issue: half of the 9 million hectares of pasture in Central America is estimated to be degraded (Szott *et al.*, 2000). Pasture degradation can be even more acute locally. For example Jansen *et al.* (1997) estimated that over 70 percent of the pastures in the Northern Atlantic zone of Costa Rica are in an advanced stage of degradation, with overgrazing and lack of sufficient N input identified as principal causes.

Main global environment concerns associated with rangeland degradation include climate change, through soil organic matter oxidation and carbon release into the atmosphere; water resources depletion through reduction of groundwater replenishment and biodiversity erosion, through habitat destruction. These issues will be further assessed in Chapter 3, 4 and 5, respectively.

2.5.3 Contamination in peri-urban environments

The ongoing geographical concentration of livestock production systems was described previously, first in peri-urban settings, then close to feed production and processing. In parallel, animal-derived food processing also locates in peri-urban areas, where the costs of transport, water, energy and services are minimized. The geographical concentration of livestock, in areas with little or no agricultural land, leads to high impacts on the environment (water, soil, air and biodiversity), mainly related to manure and waste water mismanagement. Nutrient overloads can result from several forms of mismanagement, including overfertilization of crops, overfeeding of fish ponds and improper waste disposal of agricultural (e.g. livestock) or agro-industrial wastes. Nutrient overloads coming from crop–livestock systems mainly occur when the nutrients present in manure are not properly removed or recycled. The major effects of animal waste mismanagement on the environment have been summarized by Menzi (2001) as follows:

- **Eutrophication of surface water** (deteriorating water quality, algae growth, damage to fish, etc.) owing to input of organic substances and nutrients when excreta or wastewater from livestock production get into streams through discharge, runoff or overflow of lagoons. Surface water pollution threatens aquatic ecosystems and the quality of drinking-water taken from streams. Nitrogen and phosphorus are both nutrients often associated with accelerated eutrophication of surface water (Correll, 1999; Zhang *et al*., 2003). However, phosphorus is often the limiting factor to the development of blue-green algae, which are able to utilize atmospheric N_2. Therefore, phosphorus management is often identified as a key strategy to limit surface water eutrophication from agricultural sources (Mainstone and Parr, 2002; Daniel *et al*., 1994).
- **Leaching of nitrate and possible transfer of pathogens to groundwater** from manure-storage facilities or from fields on which high doses of manure have been applied. Nitrate leaching and pathogen transfer are particular threats for drinking water quality.
- **Excess accumulation of nutrients in the soil** when high doses of manure are applied. This can threaten soil fertility owing to unbalanced or even noxious nutrient concentrations.
- **Natural areas such as wetlands and mangrove swamps are directly impacted by water pollution** often leading to biodiversity losses.

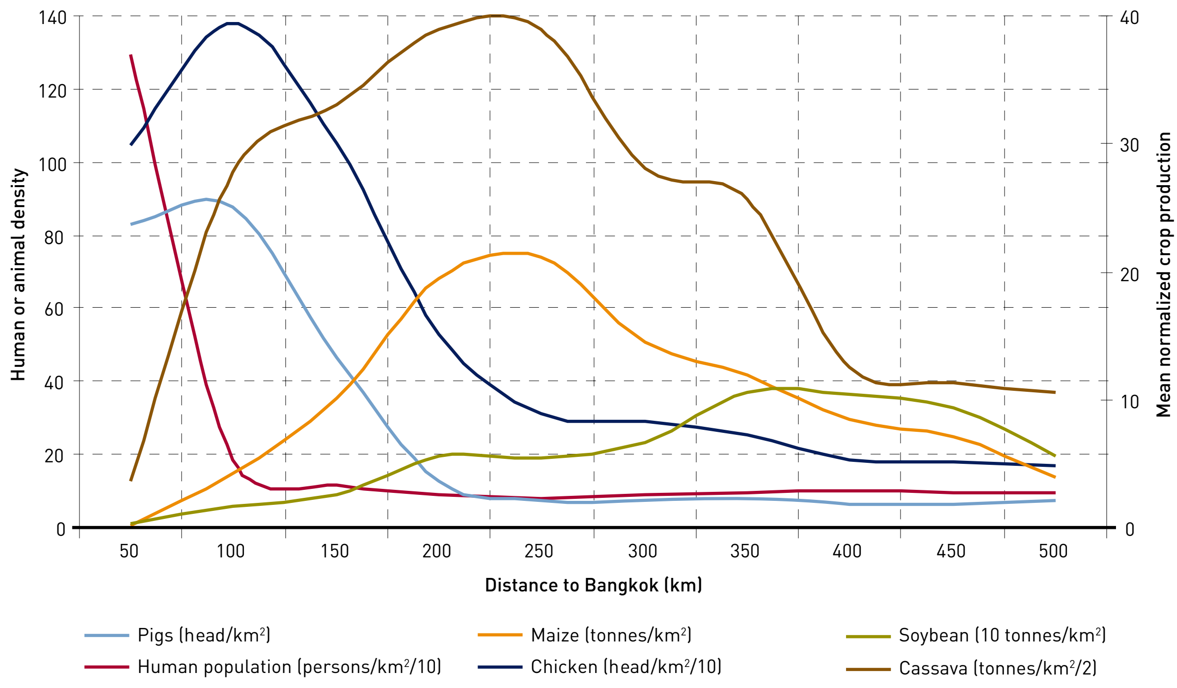
Results from LEAD studies show that in most Asian contexts, the recycling of animal manure on crops or in fish ponds (including sanitation costs) is a less expensive option than manure treatment as nutrients are removed using biochemical processes (Livestock waste management in East Asia project – LWMEA) (see Box 2.4). When production or processing units are located in peri-urban settings, far from crops and fishponds (see Figure 2.17), high transport costs make recycling practices financially unprofitable. Production units also often face high land prices and therefore tend to avoid building adequately sized treatment facilities. The result is often a direct discharge of animal manure into urban waterways, with dramatic consequences on their nutrient, drug and hormone residues and organic matter load. Manure products with high value (e.g. chicken litter, cattle dung) are, however, often marketed out of the peri-urban area.

There are also a number of animal diseases that are associated with increasing intensity of production and concentration of animals in a limited space. Many of these zoonotic diseases pose a threat to human health. Industrial and intensive forms of animal production may be a

© FAO

Farms in Prune – India, situated in proximity to apartment buildings

Figure 2.17 **Spatial distribution of humans, livestock and feed crops around Bangkok, 2001**



Source: Own calculations.

breeding ground for emerging diseases (Nipah virus, BSE), with public health consequences. Intra- and interspecies contamination risks are especially high in the peri-urban environment where high densities of humans and livestock coincide (see Figure 2.17).

As a result of economies of scale, industrial livestock production generates substantially lower income per unit of output than smallholder farms and benefits go to fewer producers. Furthermore, economic returns and spillover effects occur in the, generally, already better-off urban areas. The shift towards such production has thus, on balance, a largely negative effect on rural development (de Haan *et al.*, 2001).

Main global environment concerns associated with contamination in peri-urban environments include climate change through gaseous emissions from animal waste management, water resources depletion through pollution of surface and groundwater, and biodiversity erosion through water and soil pollution. These issues will be further assessed in Chapter 3, 4 and 5, respectively.

2.5.4 Intensive feedcrop agriculture

Crop yield improvement from intensification often has substantial environmental costs (Pingali and Heisey,1999; Tilman *et al.*, 2001). Agricultural intensification can have negative consequences at various levels:

- local: increased erosion, lower soil fertility, and reduced biodiversity;
- regional: pollution of ground water and eutrophication of rivers and lakes; and
- global: impacts on atmospheric constituents, climate and ocean waters.

Biological consequences at the agro-ecosystem level

A key aspect of intensive agriculture is the high specialization of production, often leading to monoculture with tight control of unwanted "weed" species. The reduced diversity of the plant community affects the pest complex as

Box 2.4 Livestock waste management in East Asia

Nowhere have the rapid growth of livestock production, and its impact on the environment, been more evident than in parts of Asia. During the decade of the 1990s alone, production of pigs and poultry almost doubled in China, Thailand and Viet Nam. By 2001, these three countries alone accounted for more than half the pigs and one-third of the chickens in the entire world.

Not surprisingly, these same countries have also experienced rapid increases in pollution associated with concentrations of intensive livestock production. Pig and poultry operations concentrated in coastal areas of China, Viet Nam and Thailand are emerging as the major source of nutrient pollution of the South China Sea. Along much of the densely populated coast, the pig density exceeds 100 animals per km^2 and agricultural lands are overloaded with huge nutrient surpluses (see Map 4.1, Chapter 4). Run-off is severely degrading seawater and sediment quality in one of the world's most biologically diverse shallow-water marine areas, causing "red tides" and threatening fragile coastal, marine habitats including mangroves, coral reefs and sea grasses.

The related booms in production and pollution have kindled plans for one of the most comprehensive efforts to forge an effective policy response – the Livestock Waste Management in East Asia Project (LWMEAP) – which has been prepared with the governments of China, Thailand and Viet Nam by FAO and the inter-institutional Livestock, Environment and Development Initiative (LEAD – www.lead.virtualcentre.org), under a grant from The Global Environment Facility. The project addresses environmental threats by developing policies to balance the location of livestock production operations with land resources and to encourage the use of manure and other nutrients by crop farmers. It will also set up pilot farms to demonstrate good manure management techniques.

Pollutants from all three countries threaten the South China Sea. But the nature of livestock operations differs markedly among the countries. In Thailand, three-quarters of pigs are now produced on large, industrial farms with more than 500 animals. In Viet Nam, on the other hand, very small producers with just three or four pigs account for 95 percent of production. While half of the pigs in Guangdong are still produced in operations with fewer than 100 animals, large-scale industrial operations are growing rapidly. Almost one-quarter of the pigs in Guangdong are produced on farms with more than 3 000 animals.

The LWMEAP project outlines policies at both the national and local levels. At the national level, the project stresses the need for inter-agency cooperation to develop effective and realistic regulations on environmental monitoring and manure management and to undertake spatial planning for the location of future livestock development to create the conditions for better recycling of effluents. As a key tool for shaping and implementing policy at the local level, LWMEAP provides support to the development of codes of practice adapted to the specific contexts.

Source: FAO (2004d).

well as soil invertebrates and micro-organisms, which in turn affects plant growth and health. The low diversity of monocultural agricultural systems typically results in greater crop losses from insect pests that are less diverse but more abundant (Tonhasca and Byrne, 1994; Matson *et al.*, 1997). The immediate reaction is to increase pesticide applications. As a result, pesticide diffusion along wildlife food chains and pesticide resistance has become an acute problem worldwide.

The effects of monoculture on the soil biotic community are less evident, as is effect of these changes on agro-ecosystems. Studies of key organisms however show that reduction in diversity of soil biota under agricultural practice may substantially alter the decomposition process and nutrient availability in the soil (Matson *et al.*, 1999).

Changes in natural resources

Organic matter is a critical component of soils. It provides the substrate for nutrient release, and plays a critical role in soil structure, increasing water holding capacity and reducing erosion. For intensive cropland in temperate zone agriculture, soil organic matter losses are most rapid during the first 25 years of cultivation, with typical losses of 50 percent of the original C. In tropical soils, however, such losses may occur within five years after conversion (Matson *et al.*, 1999). In addition to local impacts, the large amounts of CO_2 released in decomposition of organic matter greatly contribute to climate change.

Increasing yields also require more water. Irrigated land expanded at the rate of 2 percent per year between 1961 and 1991, and at 1 percent per year during the past decade (FAO, 2006b – see Table 1, Annex 2). This trend has dramatic consequences on the water resources. Over-pumping is a serious concern in many regions, especially where feedcrop species are cultivated outside their suitable agro-ecozone (e.g. maize in most parts of Europe), and the use of non-renewable water resources (fossil water) is frequent. Irrigation often takes place in a context of water scarcity, and this is expected to worsen as competition for withdrawals increases with human population growth, development and climate change.

Habitat deterioration

Intensification of agricultural production has been accompanied by large increases in global nitrogen (N) and phosphorus (P) fertilization. Chemical fertilizer consumption grew at 4.6 percent per year over the 1961 to 1991 period, though it stabilized thereafter (FAO, 2006b – see Table 1, Annex 2). The stabilization of fertilizer consumption at the global level results from the balance of consumption, increasing in developing countries and decreasing in developed countries.

The uptake of fertilizer nutrients by crops is limited. A significant share of P is carried away by runoff, while Matson *et al.* (1999) estimate that about 40 to 60 percent of the N that is applied to crops is left in the soil or lost by leaching. The leaching of nitrate from soils to water systems leads to increased concentrations in drinking water and contamination of ground and surface water systems, which threaten human health and natural ecosystems. In particular, eutrophication of waterways and coastal areas kills aquatic organisms and eventually causes biodiversity losses.

N fertilization, both chemical and organic, also leads to increased emissions of gases such as nitrogen oxides (NO_x), nitrous oxide (N_2O) and ammonia (NH_3). Klimont (2001) found that emissions of ammonia in China increased from 9.7 Tg in 1990 to 11.7 Tg in 1995 and are projected to rise to nearly 20 Tg NH_3 in 2030. The largest single source of emissions is the use of urea and ammonium bicarbonate - the key fertilizers in China.

Nitrogen oxide and ammonia may be transported and deposited to downwind ecosystems. This deposition can lead to soil acidification, eutrophication of natural ecosystems and shifts in species diversity, with effects on predator and parasite systems (Galloway *et al.*,1995). N deposition, mostly related to agriculture, is expected to increase dramatically over coming decades. The emission of nitrous oxides also impacts global climate, contributing to global warming - indeed the global warming potential of N_2O is 310 times greater than for carbon dioxide.

Finally, intensive agriculture land use impacts wildlife habitats. Monoculture areas offer little food or shelter to wildlife. Wild fauna is thus mostly absent from such intensive cropland.

Box 2.5 Livestock production systems and erosion in the United States

Soil erosion is regarded as one of the most important environmental problems in the United States. In the last 200 years, the United States has probably lost at least one-third of its topsoil (Barrow, 1991). Although erosion rates declined between 1991 and 2000, average erosion rates in 2001, at 12.5 tonnes per hectare per year (see Table 2.13), were still above the established sustainable soil loss rate of 11 tonnes per hectare per year (Barrow, 1991).

The rate and severity of erosion is site specific and depends largely on local conditions and soil types. However, the link with livestock production is compelling. About 7 percent of the agricultural land (2001) in the United States is devoted to the production of animal feed. Livestock production can be said to be directly or indirectly responsible for a significant proportion of the soil erosion in the United States. A careful assessment of erosion on crop and pasture lands suggests that livestock are the major contributor to soil erosion on agricultural lands, accounting for 55 percent of the total soil mass eroded every year (Table 2.13). Of this eroded mass, around 40 percent will end up in water resources. The rest will be deposited on other land sites.

Nevertheless considering the major importance of the role of agriculture land in water contamination by sediments in the United States, we can reasonably assume that livestock production systems are the major source of sediment contamination of freshwater resources.

Table 2.13

Contribution of livestock to soil erosion on agricultural lands in the United States

Erosion on cropped land	
Total erosion on cropped land *(million tonnes/year)*	1 620.8
Average water and wind cumulated erosion rate *(tonnes/ha/year)*	12.5
Total arable land for feed production *(million ha)*	51.6
Total erosion associated with feed production on cropped land *(million tonnes/year)*	**648.3**
As percentage of total erosion on cropped land	40
Erosion on pastureland	
Average water and wind cumulated erosion rate *(tonnes/ha/year)*	2
Total pastureland *(million ha)*	234
Total erosion on pastureland *(million tonnes/year)*	**524.2**
Erosion on agricultural land (crop and pasture)	
Total erosion from Agricultural land *(million tonnes/year)*	2 145.0
Total erosion associated with livestock production *(million tonnes/year)*	**1 172.5**
As percentage of total erosion on agricultural land	**55**

Source: USDA/NASS (2001); FAO (2006b).

Furthermore, intensively cropped parcels often represent a barrier to wildlife movements, leading to ecosystem fragmentation. As a consequence, Pingali and Heisey (1999) suggest that meeting the long-term demand requirements for food, and in particular cereals will require more than a shift in the yield frontier. It will also require fundamental changes in the way fertilizers and pesticides are used and soil is managed. To sustain cereal productivity growth while conserving the resource base demands that production increases should be achieved with less than proportionate increases in chemical inputs. Recent advances in fertilizer and pesticide formulae, as well as in technology and techniques for their efficient use, may help in meeting these objectives (Pingali and Heisey, 1999).

Soil erosion

Erosion rates greatly vary depending on local conditions and it is often difficult to compare local data. Erosion rates are influenced by several factors including soil structure, landscape morphology, vegetation cover, rainfall and wind levels, land use and land management including method, timing and frequency of cultivation (Stoate *et al.*, 2001) (see Box 2.5). As the worst erosion is usually caused by runoff water, erosion tends to increase as infiltration decreases. Any activity that modifies significantly the infiltration process has an impact on the erosion process.

Croplands, especially under intensive agriculture, are generally more prone to erosion than other land uses. Major factors that contribute to increased erosion rates within croplands include:

- removal of the natural vegetation that binds the soil, protects it from the wind and improves infiltration;
- inappropriate cultivation practices;
- the mechanical impact of heavy agricultural machines; and
- depletion of the natural soil fertility.

Barrow (1991) reviewed the magnitude of erosion from cropland in various countries. As the methodologies used for assessing the erosion process are not standardized it is difficult to compare the different measures. He noted that erosion levels can be extremely severe in some cases resulting in the loss of more than 500 tonnes of soil per hectare per year (observed in Ecuador and Côte d'Ivoire). As a reference, a loss of 50 tonnes per hectare per year amounts to a loss of depth of about 3 mm/yr off a soil profile. This is enough to affect agriculture in quite a short time if the top soil is shallow. There is little agreement in the literature on permissible rates of erosion but erosion levels of 0.1 to 0.2 mm per year are often considered as acceptable (Barrow, 1991).

Main global environment concerns associated with intensive feedcrop agriculture include climate change, through gaseous emissions from fertilizer applications and the decomposition of organic matter in the soil, depletion of water resources through pollution and withdrawals, and erosion of biodiversity through habitat destruction and water and soil pollution. These issues will be reviewed in Chapter 3, 4 and 5, respectively.

2.6 Conclusions

Today, the livestock sector is a major land user, spanning more than 3.9 billion hectares, representing about 30 percent of the world's surface land area. The intensity with which the sector uses land is however extremely variable. Of the 3.9 billion hectares, 0.5 are crops, generally intensively managed (Section 2.3); 1.4 are pasture with relatively high productivity and; the remaining 2.0 billion hectares are extensive pastures with relatively low productivity (Table 4, Annex 2). The sector is the first agricultural land user, accounting for about 78 percent of agricultural land and as much as 33 percent of the cropland. Despite the fact that intensive, "landless" systems have been responsible for most of the sector's growth, the influence the sector has on the cropland is still substantial, and environmental issues associated to livestock production could not be comprehensively apprehended without including the crop sector in our analysis.

As the livestock sector develops, however, its land-size requirements grow and the sector undergoes a geographical transition involving changes in land-use intensity and geographical distribution patterns.

Intensification slows the spread of livestock-related land use

The first aspect of this transition is land-use intensification. It relates to feed supply, the main purpose for which the sector uses land (either directly as pasture or indirectly as feedcrops).

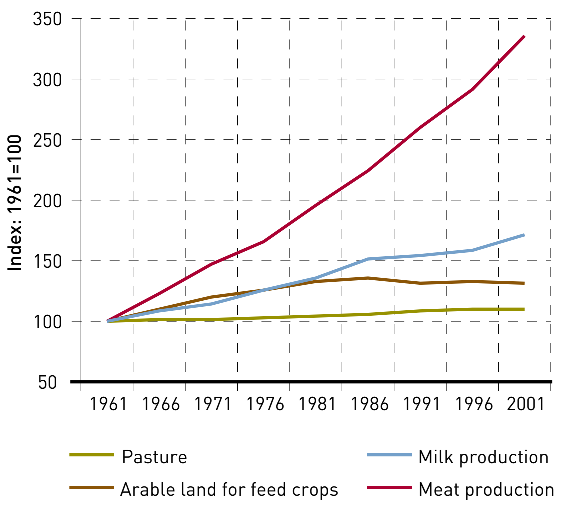
Feedcrops and cultivated pastures intensify in areas with developed transport infrastructure, strong institutions and high agro-ecological suitability. Figure 2.18 shows the marked difference in growth rates between the global areas dedicated to pasture and feed production, compared to the meat and milk outputs of the sector. This increasing productivity is the consequence of strong intensification of the sector on a global scale. The shift from ruminant species to monogastric species fed on improved diets plays a critical role in this process.

The growth in demand for livestock products will probably still play a dominant role over the next decades and lead to a net increase in the area dedicated to livestock, despite the intensification trend. Extensive pastures and feedcrop production will expand into natural habitats with low opportunity cost. It is, however, likely that the bulk of pasture and feedcrop spread has already occurred, and that the intensification process will soon overcome the trend for area expansion, leading to an eventual net decrease in the area under pasture and feedcrops.

There are regional variations to these global trends. In the EU (Figure 2.19) and more generally in OECD countries, the growth of meat and milk production happened at the same time as a reduction in the area dedicated to pasture and feedcrops. This was predominantly achieved through improved feed-conversion ratios, but part of the reduction in local feedcrop area was also compensated by feed imports, in particular from South America. Indeed, the comparable trends in South America (Figure 2.20) show a relatively stronger growth of feedcrop areas. Rapid development of a regional intensive livestock sector fuelled the feed production industry but exports were responsible for extra growth. Feedcrops grew especially rapidly in the 1970s and late 1990s, when first developed countries and then developing countries engaged in livestock industrialization and started importing protein feed.

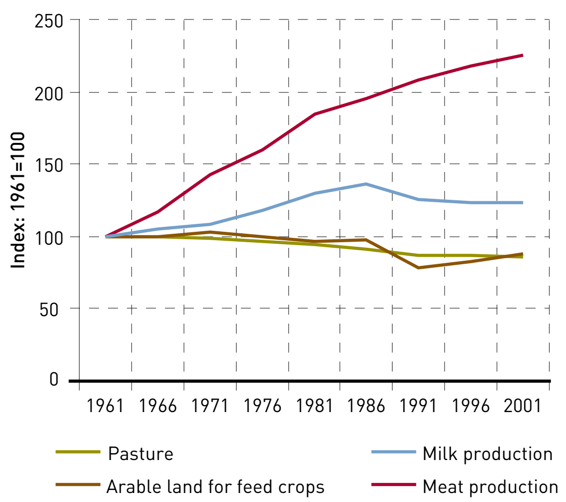
This is for example currently under way in East and Southeast Asia (Figure 2.21), where production has grown dramatically faster than the area under feedcrops and pasture (which has remained stable). This difference in growth rates has been achieved by importing feed resources,

Figure 2.18 **Global trends in land-use area for livestock production and total production of meat and milk**



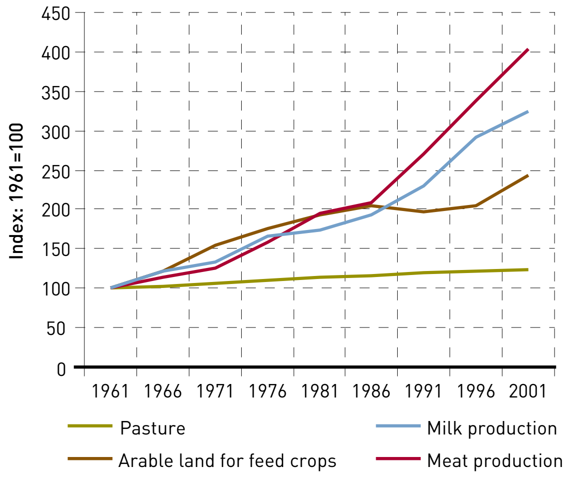
Source: FAO (2006b).

Figure 2.19 **Trends in land-use area for livestock production and local supply of meat and milk – EU-15**



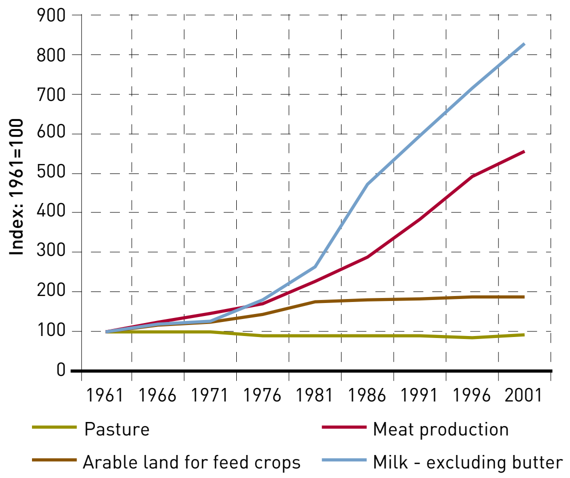
Source: FAO (2006b).

Figure 2.20 **Trends in land-use area for livestock production and local supply of meat and milk – South America**



Source: FAO (2006b).

Figure 2.21 **Trends in land-use area for livestock production and local supply of meat and milk – East and Southeast Asia (excluding China)**



Source: FAO (2006b).

and also through a rapid intensification of the livestock industry involving breed improvement, improved animal husbandry and a shift to poultry (the methodology developed to estimate land use by livestock, as well as complementary results are presented in Annex 3.1).

Production shifts to areas of feed resources or lower costs

The second feature of livestock's geographical transition lies in the changing spatial distribution of production. Production and consumption no longer coincide, as most consumption is located in urban centres, far from the feed resources. The livestock sector has adapted to this new configuration by splitting up the commodity chain and locating each specialized production or processing segment where production costs are minimized. With the development of transport infrastructure, shipment of animal products is becoming relatively cheap in comparison with other production costs. The trend towards more processed foods further contributes to reducing transport costs. Livestock production, therefore, moves closer to feed resources, or to places where the policy context (tax regime, labour standards, environmental standards), as well as access to services or disease conditions, minimize production costs. In essence, livestock are thus moving from a "default land user" strategy (i.e. as the only way to harness biomass from marginal lands, residues and interstitial areas) to an "active land user" strategy (i.e. competing with other sectors for the establishment of feed-crops, intensive pasture and production units).

Paying the environmental price

This process leads to efficiency gains in the use of resources. However, it usually develops within a context of environmental and social externalities that are mostly not addressed, and inadequate pricing of resources on the basis of private rather than social costs. As a consequence, changes in livestock geography are associated with substantial environmental impacts. For example, the private costs of transport are distortedly low and do not reflect social costs. The expansion and intensification of crop agriculture is associated with profound land degradation problems. The continuous expansion of agriculture into natural ecosystems causes climate change and biodiversity loss. The disconnection of livestock production from its feed base creates inadequate conditions for good waste management practices, which often cause soil and water pollution as well as greenhouse gas emissions.

On current trends, the ecological footprint of the livestock sector will increase because of expansion of land use and land degradation. Confronting the global environmental challenges of land use will require assessing and managing the inherent trade-offs between meeting the current demand for animal-derived foods, and maintaining the capacity of ecosystems to provide goods and services in the future (Foley *et al*., 2005). Ultimately, reaching a sustainable balance will require adequate pricing of natural resources, the internalization of externalities and the preservation of key ecosystems.

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 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).


© FAO/7398/F. BOTTS

Cracked clay soil – Tunisia 1970

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.

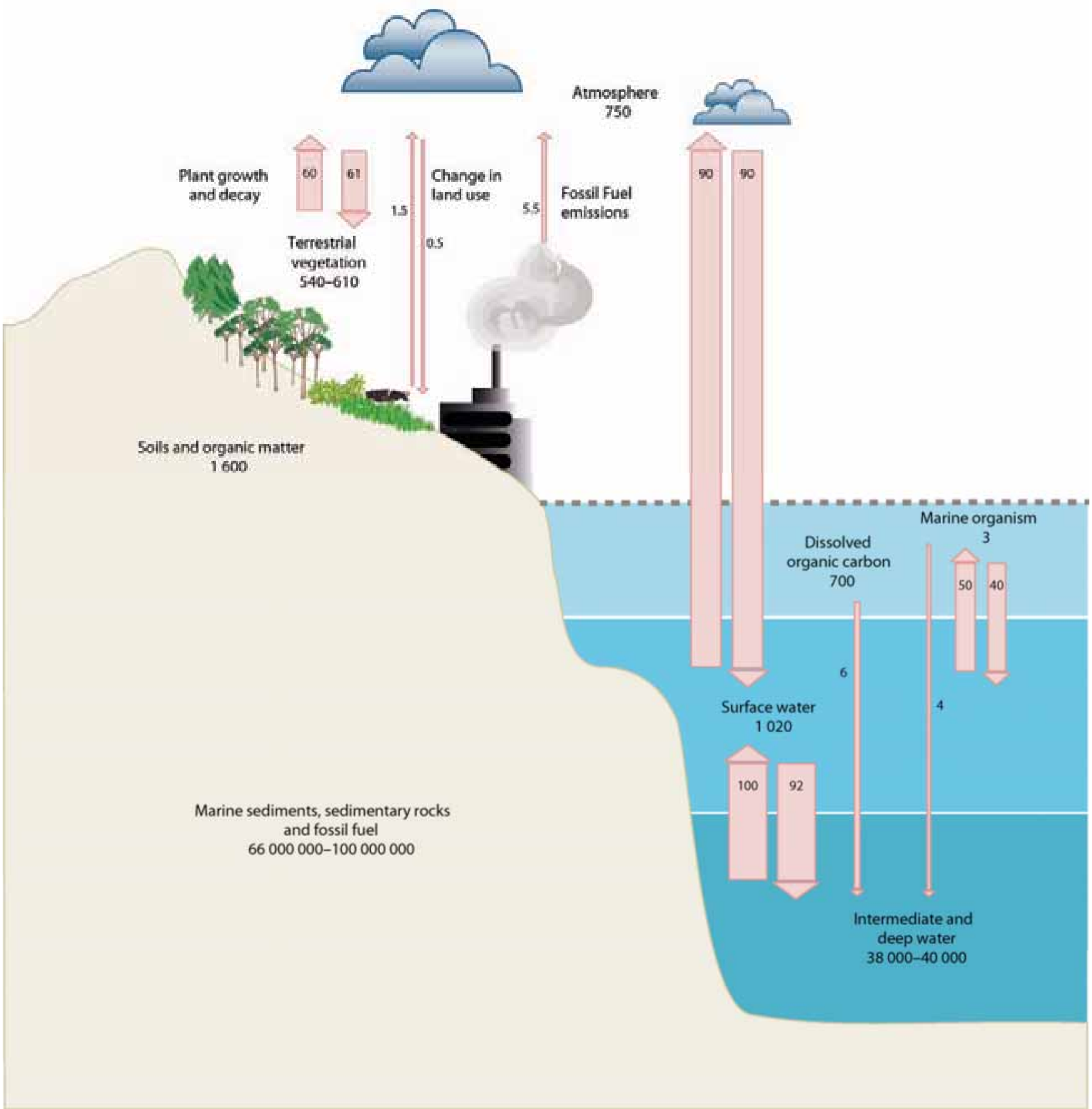
[1] The addition of substances to the atmosphere that result in direct damage to the environment, human health and quality of life is termed air pollution.

[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.

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

[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.

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.

© FAO/10460/F. BOTTS

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.

© FAO/14185/R. FAIDUTTI

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).

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

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-

Table 3.7

Global methane emissions from enteric fermentation in 2004

	Emissions (million tonnes CH_4 per year by source)					
Region/country	**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

© PHOTO COURTESY OF USDA NRCS/JEFF VANUGA

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

	Emissions (million tonnes CH_4 per year by source)						
Region/country	**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	$MJ\text{-}kg^{-1}$ live wt	Whitehead and Shupe, 1979
Eggs	6.12	$MJ\text{-}dozen^{-1}$	OECD, 1982
Pork-fresh	3.76	$MJ\text{-}kg^{-1}$ carcass	Singh, 1986
Pork-processed meats	6.30	$MJ\text{-}kg^{-1}$ meat	Singh, 1986
Sheep meat	10.4	$MJ\text{-}kg^{-1}$ carcass	McChesney *et al.*, 1982
Sheep meat-frozen	0.432	$MJ\text{-}kg^{-1}$ meat	Unklesbay and Unklesbay, 1982
Beef	4.37	$MJ\text{-}kg^{-1}$ carcass	Poulsen, 1986
Beef-frozen	0.432	$MJ\text{-}kg^{-1}$ meat	Unklesbay and Unklesbay, 1982
Milk	1.12	$MJ\text{-}kg^{-1}$	Miller, 1986
Cheese, butter, whey powder	1.49	$MJ\text{-}kg^{-1}$	Miller, 1986
Milk powder, butter	2.62	$MJ\text{-}kg^{-1}$	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).

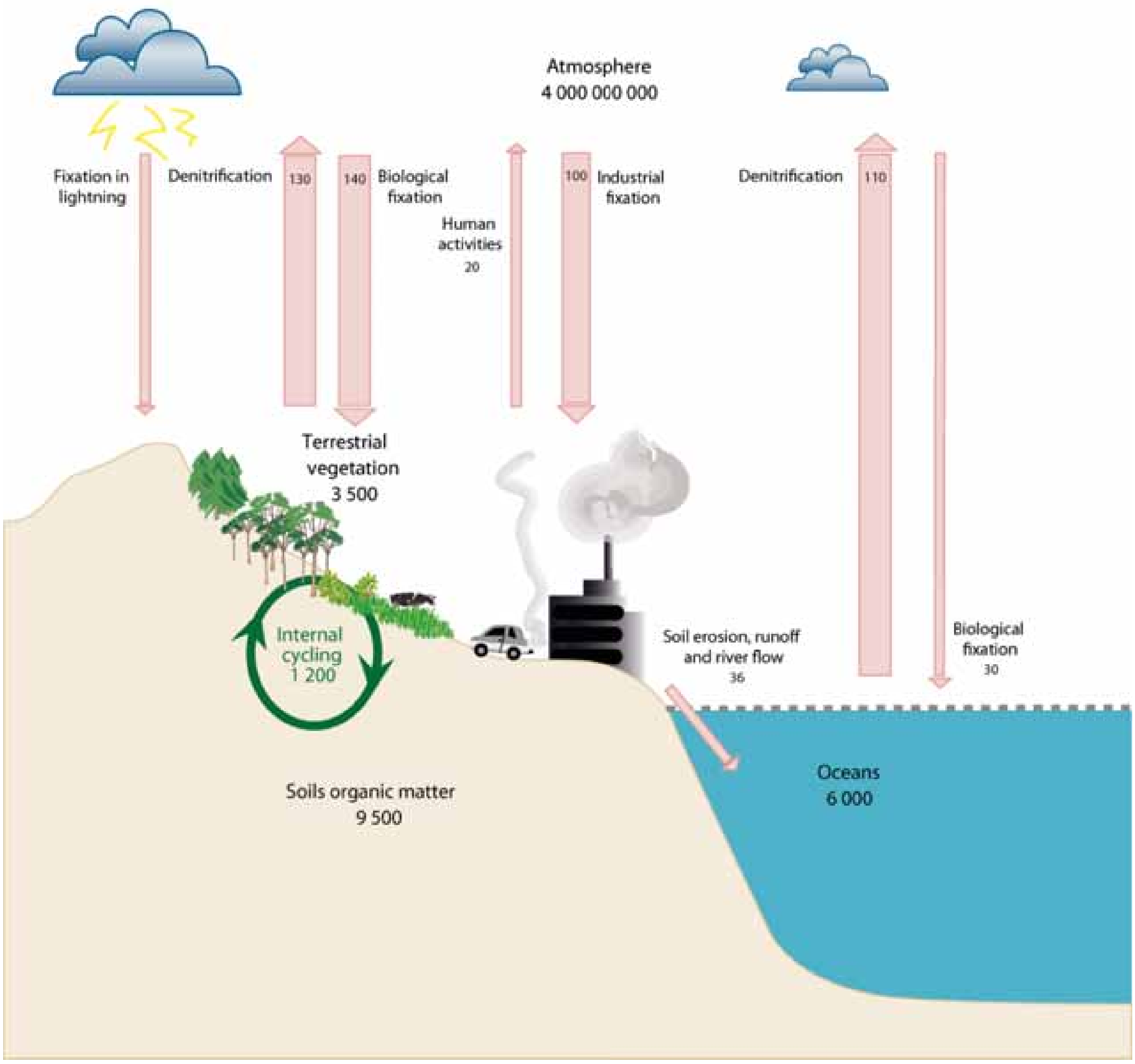
These flows represent some 60 percent of international meat trade. Annually they produce some 500 thousand tonnes of CO_2. This represents more than 60 percent of total CO_2 emissions induced by meat-related sea transport, because the trade flow selection is biased towards the long distance exchange. 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).

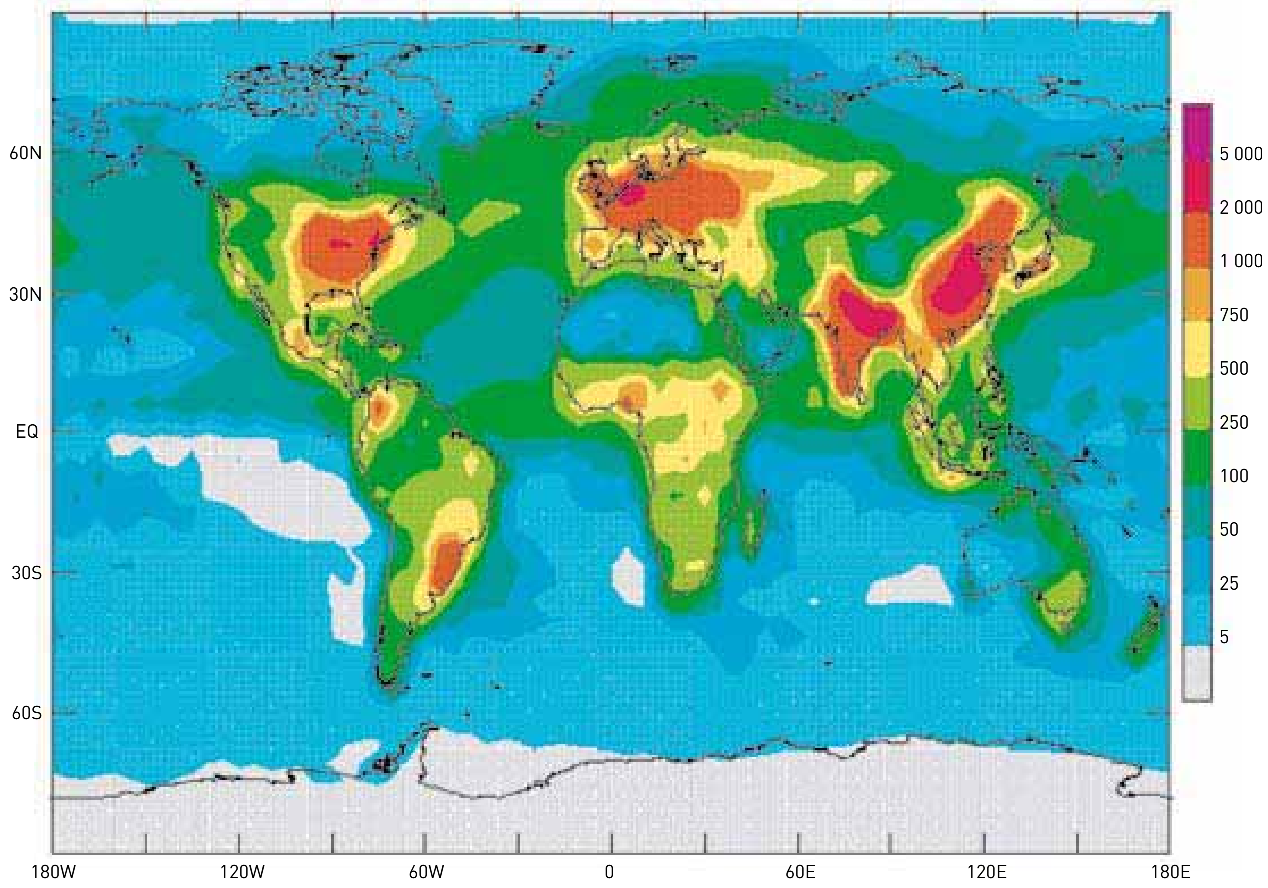
3.5 Mitigation options

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-

© LEAD/PIERRE GERBER

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.

04

Livestock's role in water depletion and pollution

4.1 Issues and trends

Water represents at least 50 percent of most living organisms and plays a key role in the functioning of the ecosystem. It is also a critical natural resource mobilized by most human activities.

It is replenished through the natural water cycle. The evaporation process, mainly from the oceans, is the primary mechanism supporting the surface-to-atmosphere portion of the cycle. Evaporation returns to ocean and water bodies as precipitation (US Geological Survey, 2005a; Xercavins and Valls, 1999).

Freshwater resources provide a wide range of goods such as drinking water, irrigation water, or water for industrial purposes, and services such as power for hydroelectricity generation and support of recreational activities to a highly diverse set of user groups. Freshwater resources are the pillar sustaining development and maintaining food security, livelihoods, industrial growth, and environmental sustainability throughout the world (Turner *et al.*, 2004).

Nevertheless, freshwater resources are scarce. Only 2.5 percent of all water resources are fresh water. The oceans account for 96.5 percent, brackish water for around 1 percent. Furthermore, 70 percent of all freshwater resources are locked up in glaciers, and permanent snow (polar caps for example) and the atmosphere (Dompka, Krchnak and Thorne, 2002; UNESCO, 2005). 110 000 km^3 of freshwater fall on earth in the form of precipitation annually, of which 70 000 km^3 evaporate immediately into the atmosphere. Out of the remaining 40 000 km^3 only 12 500 km^3 is accessible for human use (Postel, 1996).

Freshwater resources are unequally distributed at the global level. More than 2.3 billion people in 21 countries live in water-stressed basins (having between 1 000 and 1 700 m^3 per person per year). Some 1.7 billion people live in basins under scarcity conditions (with less than 1 000 m^3 per person per year) see Map 28, Annex 1 (Rosegrant, Cai and Cline, 2002; Kinje, 2001; Bernstein, 2002; Brown, 2002). More than one billion people do not have sufficient access to clean water. Much of the world's human population growth and agricultural expansion is taking place in water stressed regions.

The availability of water has always been a limiting factor to human activities, in particular agriculture, and the increasing level of demand for water is a growing concern. Excessive withdrawals, and poor water management, have resulted in lowered groundwater tables, damaged soils and reduced water quality worldwide. As a direct consequence of a lack of appropriate water resources management, a number of countries and regions are faced with ongoing depletion of water resources (Rosegrant, Cai and Cline, 2002).

Withdrawal of freshwater diverted from rivers and pumped from aquifers has been estimated at 3 906 km^3 for 1995 (Rosegrant, Cai and Cline, 2002). Part of this water returns to the ecosystem, though pollution of water resources is accelerated by the increasing discharge of wastewater into water courses. Indeed, in developing countries, 90–95 percent of public wastewater and 70 percent of industrial wastes are discharged into surface water without treatment (Bernstein, 2002).

The agricultural sector is the largest user of freshwater resources. In 2000, agriculture accounted for 70 percent of water use and 93 percent of water depletion worldwide (see Table 4.1) (Turner *et al.*, 2004). The irrigated area has multiplied nearly five times over the last century and in 2003 amounted to 277 million hectares (FAO, 2006b). Nevertheless, in recent decades, growth in the use of water resources for domestic and industrial purposes has been faster than for agriculture. Indeed, between 1950 and 1995, withdrawals for domestic and industrial uses quadrupled, while they only doubled for agricultural purposes (Rosegrant, Cai and Cline, 2002). Today people consume 30–300 litres per person a day for domestic purposes, while 3 000 litres per day are needed to grow their daily food (Turner *et al.*, 2004).

One of the major challenges in agricultural development today is to maintain food security and alleviate poverty without further depleting water resources and damaging ecosystems (Rosegrant, Cai and Cline, 2002).

The threat of increasing scarcity

Projections suggest that the situation will worsen in the next decades, possibly leading to increasing conflicts among usages and users. Under a "Business as usual scenario" (Rosegrant *et al.*, 2002), global water withdrawal is projected to increase by 22 percent to 4 772 km^3 in 2025. This increase will be driven mainly by domestic, industrial and livestock uses; the latter showing a growth of more than 50 percent. Water consumption for non-agricultural uses is projected to increase by 62 percent between 1995

Table 4.1

Water use and depletion by sector

Sector	Water use	Water depletion
	(.......... Percentages of total ....)	
Agriculture	70	93
Domestic	10	3
Industrial	20	4

Source: Brown (2002); FAO-AQUASTAT (2004).

and 2025. The use of irrigation water, however, will rise by only 4 percent over that period. The highest increase in demand for irrigation water is expected for sub-Saharan Africa and Latin America with 27 and 21 percent, respectively; both regions have only limited use of irrigation today (Rosegrant, Cai and Cline, 2002).

As a direct consequence of the expected increase in demand for water, Rosegrant, Cai and Cline (2002) projected that by 2025, 64 percent of the world's population will live in water-stressed basins (against 38 percent today). A recent International Water Management Institute (IWMI) assessment projects that by 2023, 33 percent of the world's population (1.8 billion people) will live in areas of absolute water scarcity including Pakistan, South Africa, and large parts of India and China (IWMI, 2000).

Increasing water scarcity is likely to compromise food production, as water will have to be diverted from agricultural use to environmental, industrial and domestic purposes (IWMI, 2000). Under the "business as usual scenario" mentioned above, water scarcity may cause a loss of potential production of 350 million tonnes of food, almost equal to the current total United States grain crop production (364 million tonnes in 2005) (Rosegrant, Cai and Cline, 2002; FAO, 2006b). The countries under absolute water scarcity will have to import a substantial proportion of their cereal consumption, while those unable to finance these imports will be threatened by famine and malnutrition (IWMI, 2000).

Even countries with sufficient water resources will have to expand their water supplies to make up for the increasing demand. There is widespread concern that many countries, especially in sub-Saharan Africa, will not have the required financial and technical capacity (IWMI, 2000).

Water resources are threatened in other ways. Inappropriate land use can reduce water supplies by reducing infiltration, increasing runoff and limiting the natural replenishment of groundwater resources and the maintenance of adequate stream flows, especially during dry seasons. Improper land use can severely constrain future access to water resources and may threaten the proper functioning of ecosystems. Water cycles are further affected by deforestation, an ongoing process at the pace of 9.4 million hectares per year according to FAO's latest assessment (FAO, 2005a).

Water also plays a key role in ecosystem functioning, acting as a medium and/or reactant of biochemical processes. Depletion will affect ecosystems by reducing water availability to plant and animal species, inducing a shift toward dryer ecosystems. Pollution will also harm ecosystems, as water is a vehicle for numerous pollution agents. As a result, pollutants have an impact not only locally but on various ecosystems along the water cycle, sometimes far from the initial sources.

Among the various ecosystems affected by trends in water depletion, wetlands ecosystems are especially at risk. Wetlands ecosystems are the most species-diverse habitats on earth and include lakes, floodplains, marshes and deltas. Ecosystems provide a wide range of environmental services and goods, valued globally at US$33 trillion of which US$14.9 trillion are provided by wetlands (Ramsar, 2005). These include flood control, groundwater replenishment, shoreline stabilization and storm protection, sediment and nutrient regulation, climate change mitigation, water purification, biodiversity conservation, recreation, tourism and cultural opportunities. Nevertheless, wetlands ecosystems are under great threat and are suffering from over-extraction, pollution and diversion of water resources. An estimated 50 percent of world wetlands have disappeared over the last century (IUCN, 2005; Ramsar, 2005).

The impacts of the livestock sector on water resources are often not well understood by decision-makers. The primary focus is usually the most obvious segment of the livestock commodity chain: production at farm level. But the overall water use[1] directly or indirectly by the livestock sector is often ignored. Similarly, the contribution of the livestock sector to water depletion[2] focuses mainly on water contamination by manure and waste.

This chapter attempts to provide a comprehensive overview of the livestock sector's role in the water resources depletion issue. More specifically, we will provide quantitative estimates of water use and pollution associated with the main segments of the animal food commodity chain.

We will successively also analyse livestock's contribution to the water pollution and evapo-transpiration phenomenon and its impact on the water resource replenishment process through improper land use. The final section proposes technical options for reversing these trends of water depletion.

4.2 Water use

Livestock's use of water and contribution to water depletion trends are high and growing. An increasing amount of water is needed to meet growing water requirements in the livestock production process, from feed production to product supply.

4.2.1 Drinking and servicing

Water-use for drinking and servicing animals is the most obvious demand for water resources related to livestock production. Water represents 60 to 70 percent of the body weight and is essential for animals in maintaining their vital physiological functions. Livestock meet their water requirements through drinking water, the water contained in feedstuffs and metabolic water produced by oxidation of nutrients. Water is lost from the body through respiration (lungs), evaporation (skin), defecation (intestines) and urination (kidneys). Water losses increase with high temperature and low humidity (Pallas, 1986; National Research Council, 1994, National Research Council, 1981). Reduction of water intake results in lower meat, milk and egg production. Deprivation of water quickly results in a loss of appetite and weight loss, with death occurring after a few days when the animal has lost between 15 to 30 percent of its weight.

A worker gives water to pigs raised near chicken cage on farm at Long An province – Viet Nam 2005

In extensive grazing systems, the water contained in forages contributes significantly to meeting water requirements. In dry climates, the water content of forages decreases from 90 percent during the growing season to about 10 to 15 percent during the dry season (Pallas, 1986). Air-dried feed, grains and concentrate usually distributed within industrialized production systems contain far less water: around 5 to 12 percent of feed weight (National Research Council, 2000, 1981). Metabolic water can provide up to 15 percent of water requirements.

A wide range of interrelated factors influence water needs, including: the animal species; the physiological condition of the animal; the level of

[1] "Water use" (also referred as "water withdrawals" in the literature) refers to the water removed from a source and used for human needs, some of which may be returned to the original source and reused downstream with changes in water quantity and quality. The "water demand", refers to a potential water use (adapted from Gleick, 2000).

[2] "Water depletion" (also referred as "water consumption" in the literature) refers to the use or removal of water from a water basin that renders it unavailable for other uses. It includes four generic processes: evapo-transpiration; flows to sinks; pollution; and incorporation within agricultural or industrial products (adapted from Roost *et al.*, 2003, Gleick, 2000). We deliberately chose to single out pollution in the title of this chapter, although it's covered by the notion of depletion, in order to highlight the importance of this mechanism to the reader.

Table 4.2

Drinking water requirements for livestock

Species	Physiological condition	Average Weight	Air temperature °C 15	25	35
			Water requirements		
		(kg)	*(.......... litres/animal/day*		*..........)*
Cattle	African pastoral system-lactating – 2 litres milk/day	200	21.8	25	28.7
	Large breed – Dry cows – 279 days pregnancy	680	44.1	73.2	102.3
	Large breed – Mid-lactation – 35 litres milk/day	680	102.8	114.8	126.8
Goat	Lactating – 0.2 litres milk/day	27	7.6	9.6	11.9
Sheep	Lactating – 0.4 litres milk/day	36	8.7	12.9	20.1
Camel	Mid-lactation – 4.5 litres milk/day	350	31.5	41.8	52.2
Chicken	Adult broilers (100 animals)		17.7	33.1	62
	Laying eggs (100 animals)		13.2	25.8	50.5
Swine	Lactating – daily weight gain of pigs 200g	175	17.2	28.3	46.7

Sources: Luke (2003); National Research Council (1985; 1987; 1994; 1998; 2000); Pallas (1986); Ranjhan (1998).

dry matter intake; the physical form of the diet; water availability and quality; temperature of the water offered; the ambient temperature and the production system (National Research Council, 1981; Luke, 1987). Water requirements per animal can be high, especially for highly productive animals under warm and dry conditions (see Table 4.2).

Livestock production, especially in industrialized farms, also requires service water – to clean production units, to wash animals, for cooling the facilities, the animals and their products (milk) and for waste disposal (Hutson *et al.*, 2004; Chapagain and Hoekstra, 2003). In particular, pigs require a lot of water when kept in "flushing systems [3]";in this case service water requirements can be seven times higher than drinking water needs. While data are scarce, Table 4.3 gives some indication of these water requirements. The estimates do not take into account the cooling requirements, which can be significant.

Production systems usually differ in their water use per animal and in how these requirements are met. In extensive systems, the effort expended by animals in search of feed and water increases the need for water considerably, compared to industrialized systems where animals do not move around much. By contrast, intensive production has additional service water requirements for cooling and cleaning facilities. It is also important to notice that water sourcing differs widely between industrialized and extensive production systems. In extensive livestock systems, 25 percent of the water requirements (including water services) come from feed, against only 10 percent in intensive livestock production systems (National Research Council, 1981).

In some places the importance of livestock water use for drinking and servicing compared to other sectors can be striking. For example in Botswana water use by livestock accounts for 23 percent of the total water use in the country and is the second principal user of water resources. As groundwater resources replenish only slowly, the water table in the Kalahari has substantially decreased since the nineteenth century. Other sectors will pose additional water demands in future; and water scarcity may become dramatic

[3] In a flushing system, a large volume of water carries manure down a gutter, usually sloped toward storage, such as an earthen lagoon or basin (Field *et al.*, 2001).

Table 4.3

Service water requirements for different livestock types

		Service water (litres/animal/day)	
Animal	**Age group**	**Industrial**	**Grazing**
Beef cattle	Young calves	2	0
	Adult	11	5
Dairy cattle	Calves	0	0
	Heifers	11	4
	Milking cows	22	5
Swine	Piglet	5	0
	Adult	50	25
	Lactating	125	25
Sheep	Lamb	2	0
	Adult	5	5
Goats	Kid	0	0
	Adult	5	5
Broiler chicken	Chick*100	1	1
	Adult*100	9	9
Laying hens	Chick*100	1	1
	Laying eggs*100	15	15
Horses	Foal	0	5
	Mature horses	5	5

Source: Chapagain and Hoekstra (2003).

(see Box 4.1; Els and Rowntree, 2003; Thomas, 2002). However in most countries water use for drinking and servicing remains small compared to other sectors. In the United States for example, although locally important in some states, livestock drinking and service water use was less than 1 percent of total freshwater use in 2000 (Hutson *et al.*, 2004).

Based on metabolic requirements, estimates concerning the extent of production systems and their water use, we can estimate global water use to meet livestock drinking requirements at 16.2 km^3, and service water requirements at 6.5 km^3 (not including service water requirements for small ruminants) (see Table 4.4 and 4.5). At the regional level the highest demand for servicing and drinking water is seen in South America (totalling 5.3 km^3/yr), South Asia (4.1 km^3/yr) and sub-Saharan Africa (3.1 km^3/yr). These areas represent 55 percent of global water requirements of the livestock sector.

Globally, the water requirements for livestock drinking and servicing represent only 0.6 percent of all freshwater use (see Tables 4.4 and 4.5). This direct use figure is the only one that most decision-makers take into consideration. As a result, the livestock sector is not usually considered one of the principal drivers for the depletion of freshwater resources. However, this figure is a considerable underestimate, as it does not take into account other water requirements the livestock sector entails both directly and indirectly. We will now examine the water implications of the entire production process.

4.2.2 Product processing

The livestock sector provides a wide range of commodities, from milk and meat to high value-added products such as leather or pre-cooked dishes. Going through the whole chain and identifying the share of the water use imputable to the livestock sector is a complex exercise. We focus here on the primary steps of the product processing chain, which includes slaughtering, meat and milk processing and tanning activities.

Slaughterhouses and the agro-food industry

Primary animal products such as live animals or milk, are usually processed into different meat and dairy products before consumption. Processing of meat includes a range of activities, from slaughtering to complex value-adding activities. Figure 4.1 depicts the generic process for meat, although the steps can vary depending on species. In addition to these generic processes, meat processing operations may also incorporate offal processing and rendering. Rendering converts by-products into value-added products such as tallow, meat and blood meals.

Like many other food processing activities, hygiene and quality requirements in meat processing result in high water usage and consequently high wastewater generation. Water is a major input at each processing step, except for final packaging and storage (see Figure 4.1).

Box 4.1 Livestock water use in Botswana

Predominantly a dryland country, Botswana is already experiencing 'water stress' – that is, freshwater availability ranges between 1 000 and 1 700m^3 per person per year. Livestock are a major user of freshwater resources in Botswana. In 1997, livestock accounted for 23 percent of the total water use of the country and was the second principal user of water resources (irrigation and forestry only represent 15 percent of the demand).

Groundwater resources account for 65 percent of the total water available in Botswana, but they are limited. The recharge of aquifers ranges from over 40 mm/yr in the extreme north to virtually zero in the central and western parts of the country. The rechargeable volume of groundwater for Botswana is less than 0.4 percent of Botswana's total renewable resources.

Groundwater is supplied through boreholes for domestic and livestock uses. It is estimated that there are 15 000 boreholes scattered throughout Botswana. In 1990, total water abstraction from boreholes was 76 million m^3, which was 760 percent more than the recharge rate.

Many ranches in the Kalahari have installed more boreholes than permitted in order to provide water to the increasing number of grazing animals. The increased use of boreholes has caused groundwater levels to decrease, and has probably diminished flows in natural permanent water features. As a direct consequence, the water table in the Kalahari has fallen substantially since the nineteenth century.

Under current rates of abstraction, the lifetime of surface and groundwater resources in Botswana is limited to a few decades. As water use by households is predicted to increase rapidly from approximately 29 percent (1990) to approximately 52 percent of total demand in 2020. The pressure on water resources will increase and present levels of livestock production may no longer be sustained.

Sources: Els and Rowntree (2003); Thomas (2002).

Table 4.4

Water use for drinking-water requirements

Regions	Total yearly water intake (km^3)						
	Cattle	Buffaloes	Goats	Sheep	Pigs	Poultry (100)	Total
North America	1.077	0.000	0.002	0.006	0.127	0.136	1.350
Latin America	3.524	0.014	0.037	0.077	0.124	0.184	3.960
Western Europe	0.903	0.002	0.013	0.087	0.174	0.055	1.230
Eastern Europe	0.182	0.000	0.003	0.028	0.055	0.013	0.280
Commonwealth of Independent States	0.589	0.003	0.009	0.036	0.040	0.029	0.710
West Asia and North Africa	0.732	0.073	0.140	0.365	0.000	0.118	1.430
Sub-Saharan Africa	1.760	0.000	0.251	0.281	0.035	0.104	2.430
South Asia	1.836	1.165	0.279	0.102	0.017	0.096	3.490
East and Southeast Asia	0.404	0.106	0.037	0.023	0.112	0.180	0.860
Oceania	0.390	0.000	0.001	0.107	0.010	0.009	0.520
Total	**11.400**	**1.360**	**0.770**	**1.110**	**0.690**	**0.930**	**16.260**

Sources: FAO (2006b); Luke(2003); National Research Council (1985; 1987; 1994; 1998; 2000a); Pallas (1986); Ranjhan (1998).

Table 4.5

Water use for service water requirements

Region	Service water (km^3)			
	Cattle	Pigs	Poultry (100)	Total
North America	0.202	0.682	0.008	0.892
Latin America	0.695	0.647	0.009	1.351
Western Europe	0.149	1.139	0.004	1.292
Eastern Europe	0.028	0.365	0.001	0.394
Commonwealth of Independent States	0.101	0.255	0.002	0.359
West Asia and North Africa	0.145	0.005	0.006	0.156
Sub-Saharan Africa	0.415	0.208	0.003	0.626
South Asia	0.445	0.139	0.003	0.586
East and Southeast Asia	0.083	0.673	0.009	0.765
Oceania	0.070	0.051	0.000	0.121
Total	**2.333**	**4.163**	**0.046**	**6.542**

Note: Calculation based on Chapagain and Hoekstra (2003).

At red meat (beef and buffalo) abattoirs, water is used primarily for washing carcasses at various stages and for cleaning. Of total water use for processing, between 44 and 60 percent is consumed in the slaughter, evisceration and boning areas (MRC, 1995). Water usage rates range from 6 to 15 litres per kilo of carcass. Given that the world production of beef and buffalo meat was 63 million tonnes in 2005 a conservative estimate of the water use for these stages would lie between 0.4 and 0.95 km^3, i.e. between 0.010 percent and 0.024 percent of global water use (FAO, 2005f).

In poultry processing plants, water is used to wash carcasses and cleaning; hot water scalding of birds prior to defeathering; in water flumes for transporting feathers, heads, feet and viscera and for chilling birds. Poultry processing tends to be more water-intensive per weight unit than red meat processing (Wardrop Engineering, 1998). Water use is in the range 1 590 litres per bird processed (Hrudey, 1984). In 2005, a total of 48 billion birds were slaughtered globally. A conservative estimate of global water use would be around 1.9 km^3, representing 0.05 percent of the water use.

Dairy products also require significant amounts of water. Best practice water use in commercial milk processes is reported to be 0.8 to 1 litre water/kg of milk (UNEP, 1997a). These conservative estimates result in a global water use for milk processing over 0.6 km^3 (0.015 percent of the global water use), not considering water used for derived products, especially cheese.

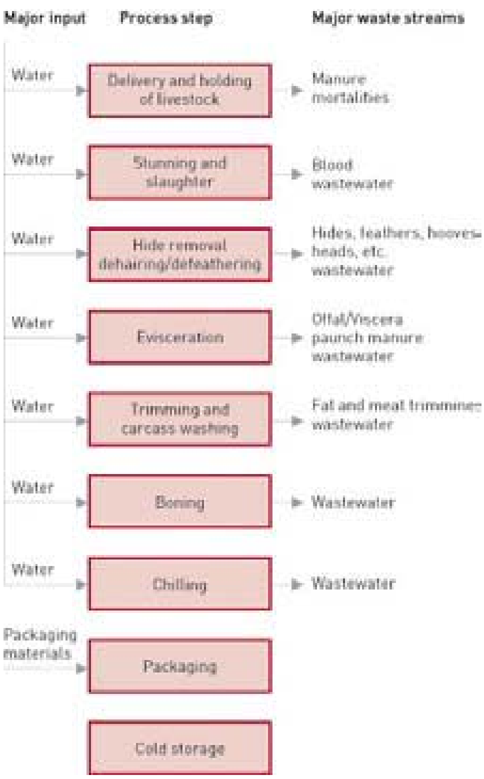
Tanneries

Between 1994 and 1996 approximately 5.5 million tonnes of raw hides were processed each year to produce 0.46 million tonnes of heavy leather and about 940 million m^2 of light leather. A further 0.62 million tonnes of raw skins on a dry basis were converted into almost 385 million m^2 of sheep and goat leather.

The tanning process includes four main operational steps: storage and beam house; tanyard; post tanning; and finishing. Depending on the type of technology applied, the water requirements for processing skins vary greatly, from 37 to 59 m^3 per tonne of raw hides when using conventional technologies to 14 m^3 when using advanced technologies (see Table 4.6). This amounts to a world total of 0.2 to 0.3 km^3 per year (0.008 percent of global water use).

The water use requirements for processing

Figure 4.1 **Flow diagram for meat processing operations**



Source: UNEP (2004a).

animal products can have a significant environmental impact in some locations. However, the main environmental threat lies in the volume of pollutants discharged locally by the processing units.

4.2.3 Feed production

As previously described, the livestock sector is the world's largest anthropogenic land user. The vast majority of this land, and much of the water it contains and receives are destined for feed production.

Evapotranspiration is the main mechanism by which crop and grassland deplete water

Table 4.6

Water use and depletion in tanning operations

Operation	Discharge (m^3/tonne raw hide) Conventional technology	Advanced technology
Soaking	7–9	2
Liming	9–15	4.5
Deliming, bating	7–11	2
Tanning	3–5	0.5
Post-Tanning	7–13	3
Finishing	1–3	0
Total	**34–56**	**12**

Source: Gate information services – GTZ (2002)

resources. When water, evapotranspired by feed cropland, is attributed to the production of livestock, the amounts involved are so large that the other water uses described above pale by comparison. Zimmer and Renault (2003) for example show in a rough accounting effort that the livestock sector may account for some 45 percent of the global budget of water used in food production. However, a large share of this water use is not environmentally significant. Evapotranspiration by grasslands and non-cultivated fodder land used for grazing represents a large share. This water generally has little to no opportunity cost, and indeed the amount of water lost in the absence of grazing might not

Handline sprinkler irrigation system – United States 2000 be any lower. More intensively managed grazing lands often have agricultural potential, but are mostly located in water abundant areas, i.e. here it is more the land that has the opportunity cost rather than water.

Water used for feed production in extensive land-based livestock production systems is not expected to substantially increase. As stated previously, grazing systems are in relative decline in most parts of the world. One important reason is that most grazing is in arid or semi-arid zones where water is scarce, limiting the expansion or intensification of livestock production. Production from mixed systems is still expanding rapidly, and water is not a limiting factor in most situations. Here, productivity gains are expected from an increased level of integration between livestock and crop production, with animals consuming considerable amounts of crop residues.

In contrast, more intensively managed mixed systems and industrial livestock systems are characterized by a high level of external inputs, i.e. concentrate feed and additives, often transported over long distances. The demand for these products, and thereby demand for the corresponding raw materials (i.e. cereal and oil crops), is increasing rapidly[4]. In addition, cereal and oil crops occupy arable land, where water generally has a considerable opportunity cost. Substantial amounts are produced by irrigation in relatively water short areas[5]. In such areas the livestock sector may be directly responsible for severe environmental degradation through water depletion, depending on the source of the irrigation water. Although, in rainfed areas, even the increasing appropriation of arable land by the sector may, more indirectly, lead to depletion of available water because it reduces the water available for other uses, particularly food crops.

In view of the increase of "costly" water use by the livestock sector, it is important to assess its current significance. Annex 3.4 presents a methodology for quantifying this type of livestock water use and assessing its significance. This assessment is based on spatially detailed water-balance calculations and information available for the four most important feedcrops: barley, maize, wheat and soybean (hereafter referred to as BMWS). The results presented in Table 4.7, therefore, do not represent the entire feed crop water use. These four crops account for roughly three-quarters of the total feed used in the intensive production of monogastrics. For other significant users of these external inputs, i.e. the intensive dairy sector, this share is in the same order of magnitude.

Annex 3.4 describes two different approaches that are designed to deal with uncertainty in estimating water use by feed crops, related to lack of knowledge of the locations of feed-dedicated cropping. As Table 4.7 shows, these two

[4] An increasing share of the increment in the production of cereals, mainly coarse grains, will be used in livestock feed. As a result, maize production in the developing countries is projected to grow at 2.2 percent p.a. against «only» 1.3 percent for wheat and 1.0 percent for rice (FAO, 2003a). Such contrasts are particularly marked in China where wheat and rice production is expected to grow only marginally over the projection period of aforementioned report, while maize production is expected to nearly double.

[5] FAO (2003a) estimates that about 80 percent of the projected growth in crop production in developing countries will come from intensification in the form of yield increases (67 percent) and higher cropping intensities (12 percent). The share due to intensification will go up to 90 percent and higher in the land-scarce regions of the West Asia/North Africa and South Asia. It is estimated that in the developing countries at present, irrigated agriculture, with about a fifth of all arable land, accounts for 40 percent of all crop production and almost 60 percent of cereal production. The area equipped for irrigation in developing countries is projected to expand by 40 million hectares (20 percent) over the projection period. This underlines the importance of the livestock sector's responsibility for irrigation water use: feed production may intensify in many locations, but particularly production hot spots like central China, the mid west of the United States, and the Latin American area covered by Eastern Paraguay, Southern Brazil and Northern Argentina may develop into increasingly important global centres of supply that will both expand and intensify, which may turn currently sufficient water supply levels into a limiting production factor.

approaches yield very similar results. This suggests that despite a certain number of unverified assumptions, the resulting aggregate quantities may provide fairly accurate estimates.

Globally, BMWS feed accounts for some 9 percent of all irrigation water evapotranspired globally. When we include evapotranspiration of water received from precipitation in irrigated areas, this share rises to some 10 percent of total water evapotranspired in irrigated areas. Considering that BMWS unprocessed feed material represents only some three-quarters of the feed given to intensively managed livestock, nearly 15 percent of water evapotranspired in irrigated areas can probably be attributed to livestock.

There are pronounced regional differences. In sub-Saharan Africa and in Oceania, very little irrigation is dedicated to BMWS feed, either in absolute or in relative terms. In South Asia/India, the amount of irrigation water evapotranspired by BMWS feed, although considerable, represents only a small share of total water evapotranspired through irrigation. Similar absolute amounts in the more water short West Asia and North Africa region represent some 15 percent of total water evapotranspired in irrigated areas. By far the highest share of water evapotranspired through irrigation is found in Western Europe (over 25 percent), followed by eastern Europe (some 20 percent). Irrigation is not very widespread in Europe, which is generally not short of water, and indeed the corresponding BMWS feed irrigation water use is less in absolute terms than for WANA. But the southern part of Western Europe regularly suffers summer droughts. In southwestern France for example irrigated maize (for feed) has repeatedly been held responsible for severe drops in the flow of major rivers, as well as damage to coastal aquaculture during such summer droughts, and unproductive pastures for the ruminant sector (Le Monde, 31-07-05). The highest absolute quantities of BMWS feed irrigation water evapotranspired are found in the United States and in East and Southeast Asia (ESEA), in both cases also representing a high share of the total (about 15 percent). A considerable portion of the irrigation water in the United States originates from fossil groundwater resources (US Geological Survey, 2005). In ESEA, in view of the changes under way in the livestock sector, water depletion and conflicts over its use may become serious problems over the coming decades.

Despite its environmental relevance, irrigation water represents only a small part of total BMWS feed water evapotranspired (6 percent globally). With respect to other crops, BMWS feed in North and Latin America is preferentially located in rainfed areas: its share in rainfed evapotranspiration is much larger than that in the evapotranspiration of irrigation water. In Europe on the contrary BMWS feed is preferentially irrigated, while even in a critically water-short region such as WANA, the BMWS feed share of evapotranspiration from irrigated land exceeds that of rainfed arable land. It is clear that feed production consumes large amounts of critically important water resources and competes with other usages and users.

4.3 Water pollution

Most of the water used by livestock returns to the environment. Part of it may be re-usable in the same basin, while another may be polluted[6] or evapotranspired and, thereby, depleted. Water polluted by livestock production, feed production and product processing detracts from the water supply and adds to depletion.

Pollution mechanisms can be separated into point source and non–point source. Point-source pollution is an observable, specific and confined discharge of pollutants into a water body. Applied to livestock production systems, point-

[6] Water pollution is an alteration of the water quality by waste to a degree that affects its potential use and results in modified physico-chemical and microbiological properties (Melvin, 1995).

Table 4.7

Evapotranspiration of water for production of barley, maize, wheat and soybeanbean (BMWS) for feed

Region/Country	Irrigated BMWS feed			Rainfed BMWS feed		BMWS feed irrigation water ET as percentage of total BMWS feed water ET
	Evapotranspired irrigation water km³	Percentage of total irrigation water evapotranspired	Percentage of total water evapotranspired in irrigated areas[1]	Water evapotranspired km³	Percentage of total water evapotranspired in rainfed cropland	
North America	**14.1** – 20.0	9 – 13	11 – 15	321 – **336**	21 – 22	4 – 6
Latin America and the Caribbean	3.0 – **3.8**	6 – 8	7 – 9	220 – **282**	12 – 15	1
Western Europe	**8.5** – 9.5	25 – 28	25 – 29	**65** – 99	14 – 22	7 – 10
Eastern Europe	**1.8** – 2.4	17 – 22	19 – 23	**30** – 46	12 – 18	4 – 5
Commonwealth of Independent States	**2.3** – 6.0	3 – 7	3 – 7	**19** – 77	2 – 8	7 – 9
West Asia and North Africa	**11.2** – 13.1	9 – 10	13 – 14	**30** – 36	9 – 11	17 – 19
Sub-Saharan Africa	0.2	1	1	**20** – 27	1 – 2	1
South Asia	**9.1** – 11.7	2 – 3	2 – 3	**36** – 39	3	16 – 18
East and Southeast Asia	20.3 – **30.1**	14 – 20	13 – 18	226 – **332**	11 – 16	6 – 7
Oceania	**0.3** – 0.6	3 – 5	3 – 5	**1.7** – 12	1 – 4	5 – 12
Australia	**0.3** – 0.6	3 – 5	4 – 6	**1.4** – 11	1 – 5	5 – 14
China	15.3 – **19.3**	14 – 18	15 – 16	141 – **166**	14 – 16	7 – 8
India	**7.3** – 10.0	3	2 – 3	**30** – 36	3	17 – 18
Brazil	0.2 – **0.4**	6 – 10	9 – 14	123 – **148**	14 – 16	0
World	**81** – 87	8 – 9	10	1 103 – **1 150**	10 – 11	6

Note: Figures in bold represent results of the Spatial Concentration approach. Other figures are based on the area wide integration approach (see Annex 3.4 for details on the methodology). All figures are actual evapotranspiration (ET) estimates, based on total irrigation and natural ET data provided by J. Hoogeveen, FAO (estimated according to the methodology described in FAO, 2003a).

[1] Evapotranspiration from irrigated areas is the sum of evapotranspiration from irrigation water and evapotranspiration from precipitation in irrigated areas.

Source: Own calculations.

source pollution refers to feedlots, food processing plants, and agrichemical processing plants. Non-point source pollution is characterized by a diffuse discharge of pollutants, generally over large areas such as pastures.

4.3.1 Livestock waste

Most of the water used for livestock drinking and servicing returns to the environment in the form of manure and wastewater. Livestock excreta contain a considerable amount of nutrients (nitrogen, phosphorous, potassium), drug residues, heavy metals and pathogens. If these get into the water or accumulate in the soil, they can pose serious threats to the environment (Gerber and Menzi, 2005). Different mechanisms can be involved in the contamination of freshwater resources by manure and wastewater. Water contamination can be direct through the loss via runoff from farm buildings, losses from failure of storage facilities, deposition of faecal material into freshwater sources and deep percolation and transport through soil layers via drainage waters at farm level. It can also be indirect through non-point source pollution from surface runoff and overland flow from grazing areas and croplands.

Table 4.8

Nutrient intake and excretions by different animals

Animal	Intake (kg/year)		Retention (kg/year)		Excretion (kg/year)		Percentage of N excreted in mineral form[1]
	N	P	N	P	N	P	
Dairy cow[2]	163.7	22.6	34.1	5.9	129.6	16.7	69
Dairy cow[3]	39.1	6.7	3.2	0.6	35.8	6.1	50
Sow[2]	46.0	11.0	14.0	3.0	32.0	8.0	73
Sow[3]	18.3	5.4	3.2	0.7	15.1	4.7	64
Growing pig[2]	20.0	3.9	6.0	1.3	14.0	2.5	78
Growing pig[3]	9.8	2.9	2.7	0.6	7.1	2.3	59
Layer hen[2]	1.2	0.3	0.4	0.0	0.9	0.2	82
Layer hen[3]	0.6	0.2	0.1	0.0	0.5	0.1	70
Broiler[2]	1.1	0.2	0.5	0.1	0.6	0.1	83
Broiler[3]	0.4	0.1	0.1	0.0	0.3	0.1	60

[1] Assumed equivalent to urine N excretion. As mineral N is susceptible to volatilization, this percentage is often lower in manure applied on the land.
[2] Highly productive situations
[3] Less productive situations.
Note: Owing to the variation in intake and nutrient content of the feeds, these values represent examples, not averages, for highly and less productive situations.
Source: de Wit *et al.*, (1997).

The main pollutants

Nutrient surpluses stimulate eutrophication and may represent a health hazard

Nutrient intake by animals can be extremely high (see Table 4.8). For example a productive dairy cow ingests up to 163.7 kg of N and 22.6 kg of P per year. Some of the nutrients ingested are sequestered in the animal, but most of it return to the environment and may represent a threat to water quality. Annual nutrient excretions by different animals are presented in Table 4.8. In the case of a productive dairy cow 129.6 kg of N (79 percent of the total ingested) and 16.7 kg of P (73 percent) is excreted every year (de Wit *et al.*, 1997). The phosphorus load excreted by one cow is equivalent to that of 18–20 humans (Novotny *et al.*, 1989). Nitrogen concentration is highest in hog manure (76.2 g/N/kg dry weight), followed by turkeys (59.6 g/kg), poultry layers (49.0), sheep (44.4), poultry broilers (40.0), dairy cattle (39.6) and beef cattle (32.5). Phosphorus content is highest in poultry layers (20.8 g/P/kg dry weight), followed by hogs (17.6), turkeys (16.5), poultry broilers (16.9), sheep (10.3), beef (9.6) and dairy cattle (6.7) (Sharpley *et al.*, 1998 in Miller, 2001). In intensive production areas, these figures result in high nutrient surpluses that can overwhelm the absorption capacities of local ecosystems and degrade surface and groundwater quality (Hooda *et al.*, 2000).

According to our assessment, at the global level, livestock excreta in 2004 were estimated to contain 135 million tonnes of N and 58 million tonnes of P. In 2004, cattle were the largest contributors for the excretion of nutrients with 58 percent of N; pigs accounted for 12 percent and poultry for 7 percent.

The major contributors of nutrients are the mixed production systems that represent 70.5 percent of N and P excretion, followed by grazing systems with 22.5 percent of the annual N and P excretion. Geographically the biggest single contributor is Asia, which represents 35.5 percent of the global annual excretion of N and P.

High concentrations of nutrients in water resources can lead to over-stimulation of aquatic plant and algae growth leading to eutrophication, undesirable water flavour and odour, and excessive bacterial growth in distribution systems. They can protect micro-organisms from the effect of salinity and temperature, and may pose a public health hazard. Eutrophication is a natural process in the ageing of lakes and some estuaries, but livestock and other agriculture-related activities can greatly accelerate eutrophication by increasing the rate at which nutrients and organic substances enter aquatic ecosystems from their surrounding watersheds (Carney *et al.*, 1975; Nelson *et al.*, 1996). Globally, the deposition of nutrients (especially N) exceeds the critical loads for eutrophication for 7–18 percent of the area of natural and semi-natural ecosystems (Bouwman and van Vuuren, 1999).

If the plant growth resulting from eutrophication is moderate, it may provide a food base for the aquatic community. If it is excessive, algal blooms and microbial activity may overuse dissolved oxygen resources, which can damage the proper functioning of ecosystems. Other adverse effects of eutrophication include:

- shifts in habitat characteristics owing to change in the mix of aquatic plants;
- replacement of desirable fish by less desirable species, and the associated economic losses;
- production of toxins by certain algae;
- increased operating expenses of public water supplies;
- infilling and clogging of irrigation canals with aquatic weeds;
- loss of recreational use opportunities; and
- impediments to navigation due to dense weed growth.

These impacts occur both in freshwater and marine ecosytems, where algal blooms are reported to cause widespread problems by releasing toxins and causing anoxia ("dead zones"), with severe negative impacts on aquaculture and fisheries (Environmental Protection Agency, 2005; Belsky, Matze and Uselman, 1999; Ongley, 1996; Carpenter *et al.*, 1998).

Phosphorus is often considered as the key limiting nutrient in most aquatic ecosystems. In proper functioning ecosystems the ability of wetlands and streams to retain P is then crucial for downstream water quality. But an increasing number of studies have identified N as the key limiting nutrient. In general terms, P tends to be more of a problem with surface water quality, whereas N tends to pose more of a threat to groundwater quality by nitrate leaching through soil layers (Mosley *et al.*, 1997; Melvin, 1995; Reddy *et al.*, 1999; Miller, 2001; Carney, Carty and Colwell, 1975; Nelson, Cotsaris and Oades, 1996).

Nitrogen: Nitrogen is present in the environment in different forms. Some forms are harmless, while others are extremely harmful. Depending on its form, N can be stored and immobilized within the soil, or it can leach to groundwater resources, or it can be volatized. Inorganic N is very mobile through the soil layers compared to organic N.

Nitrogen is excreted by livestock both in organic and inorganic compounds. The inorganic fraction is equivalent to the N emitted in urine and is usually greater than the organic one. Direct losses of N from excreta and manure take four main forms: ammonia (NH_3), dinitrogen (N_2), nitrous oxide (N_2O) or nitrate (NO_3-) (Milchunas and Lauenroth, 1993; Whitmore, 2000). Part of the inorganic N is volatized and emitted as ammonia in animal houses, during deposition and manure storage, after manure application and on pastures.

Storage and application conditions of manure greatly influence the biological transformation of the N compounds, and the resulting compounds pose different threats to the environment. Under anaerobic conditions, nitrate is transformed into harmless N_2 (denitrification). However, if organic carbon is deficient, relative to nitrate, the production of the harmful by-product N_2O increases. This suboptimal nitrification occurs when ammonia is washed directly from the soil into the water resources (Whitmore, 2000; Carpenter *et al.*, 1998).

Leaching is another mechanism whereby N is lost to water resources. In its nitrate (NO_3) form (inorganic N), nitrogen is very mobile in soil solution, and can easily be leached below the rooting zone to the groundwater, or enter the subsurface flow. Nitrogen (especially its organic forms) can also be carried into water systems through runoff. The high levels of nitrate observed in water courses close to grazing areas are mainly the result of groundwater discharges and subsurface flow. When manure is used, as an organic fertilizer, much of the nitrogen losses after application are associated with mineralization of soil organic matter at a time when there is no crop cover (Gerber and Menzi, 2005; Stoate *et al.*, 2001; Hooda *et al.*, 2000).

High levels of nitrate within water resources may represent a health hazard. Excessive levels in drinking water may cause methemoglobinemia ("blue baby syndrome") and can poison human infants. Among adults, nitrate toxicity may also cause abortion and stomach cancers. The WHO guide value for nitrate concentration in drinking water is 45 mg/litre (10 mg/litre for NO_3-N) (Osterberg and Wallinga, 2004; Bellows, 2001; Hooda *et al.*, 2000). Nitrite (NO_2-) is just as susceptible to leaching as nitrate, and is far more toxic.

The serious water pollution threat represented by industrialized livestock production systems has been widely described. In the United States, for example, Ritter and Chirnside (1987) analysed NO_3-N concentration in 200 groundwater wells in Delaware (cited in Hooda *et al.*, 2000). Their results demonstrated the high local risk presented by industrial livestock production systems: in poultry production areas, the mean concentration rate was 21.9 mg/litres compared to 6.2 for corn production areas and 0.58 for forested areas. In another study in Southwest Wales (United Kingdom), Schofield, Seager and Merriman, (1990) show that a river draining exclusively from livestock farming areas was heavily polluted with background levels of 3-5 mg/litres of NH_3-N and peaks as high as 20 mg/litres. The high peaks may be after rains, because of waste washing from farm backyards and manured fields (Hooda *et al.*, 2000).

Similarly, in Southeast Asia the LEAD initiative analysed the land-based sources of pollution to the South China Sea, with particular emphasis on the contribution of the growing swine industry in China, Thailand, Viet Nam and China's Guangdong province. Pig waste was estimated to be a greater contributor to pollution than human domestic sources in the three countries. The share of nutrient emissions in water systems attributable to pig waste ranges from 14 per-

Table 4.9

Estimated relative contribution of pig waste, domestic wastewater and non-point sources to nitrogen and phosphorus emissions in water systems

Country/Province	Nutrient	Potential load *(tonnes)*	Percentage contribution to nutrient emissions in water systems: Pig waste	Domestic wastewater	Non-point source
China-Guangdong	N	530 434	72	9	19
	P	219 824	94	1	5
Thailand	N	491 262	14	9	77
	P	52 795	61	16	23
Viet Nam	N	442 022	38	12	50
	P	212 120	92	5	3

Source: FAO (2004d).

cent for N and 61 percent for P in Thailand to 72 percent for N and 94 percent for P in the China's Guangdong province (see Table 4.9) (Gerber and Menzi, 2005).

Phosphorus: Phosphorus in water is not considered to be directly toxic to humans and animals and, therefore, no drinking-water standards have been established for P. Phosphorus contaminates water resources when manure is directly deposited or discharged into the stream or when excessive levels of phosphorus are applied to the soil. Unlike nitrogen, phosphorus is held by soil particles and is less subject to leaching unless concentration levels are excessive. Erosion is in fact the main source of phosphate loss and phosphorus is transported in surface runoff in soluble or particulate forms. In areas with high livestock densities phosphorus levels may build up in soils and reach water courses through runoff. In grazing systems cattle treading on soil affects the infiltration rate and macroporosity, and causes loss of sediment and phosphorus via overland flow from pasture and cultivated soil (Carpenter *et al*., 1998; Bellows, 2001; Stoate *et al*., 2001; McDowell *et al*., 2003).

Total organic carbon reduces oxygen levels in water

Organic waste generally contains a large proportion of solids with organic compounds that can threaten water quality. Organic contamination may stimulate proliferation of algae, which increases their demand for oxygen and reduces available oxygen for other species. The biological oxygen demand (BOD) is the indicator usually used to reflect water contamination by organic materials. A literature review by Khaleel and Shearer, (1998) found a strong correlation between high BOD and high livestock numbers or the direct discharge of farm effluents. Rain plays a major role in the variation of BOD levels in streams draining livestock areas, unless farm effluents are directly discharged into the stream (Hooda *et al*., 2000).

Table 4.10

Ranges of BOD concentration for various wastes and animal products

Source	BOD (mg/litre)
Milk	140 000
Silage effluents	30 000–80 000
Pig slurry	20 000–30 000
Cattle slurry	10 000–20 000
Liquid effluents draining from slurry stores	1 000–12 000
Dilute dairy parlour and yard washing (dirty water)	1 000–5 000
Untreated domestic sewage	300
Treated domestic sewage	20–60
Clean river water	5

Source: MAFF–UK (1998).

Table 4.10 presents the BOD levels for various wastes in England. Livestock-related wastes are among those with the highest BOD. The impacts of total organic carbon and associated levels of BOD on water quality and on the ecosystems have been assessed at the local level but lack of data make extrapolation at higher scales impossible.

Biological contamination represents a public health hazard

Livestock excrete many zoonotic micro-organisms and multi-cellular parasites of relevance to human health (Muirhead *et al*., 2004). Pathogenic micro-organisms can be water-borne or food-borne, especially if the food crops are watered with contaminated water (Atwill, 1995). High quantities of pathogens have usually to be directly discharged for an effective transmission process to occur. Several biological contaminants can survive for days and sometimes weeks in the faeces applied on land and may later contaminate water resources via runoff.

The most important water-borne **bacterial and viral pathogens** that are of primary importance to public health and veterinary public health are:

Campylobacter spp.: Various species of *campylobacter* have an important role in human gastrointestinal infection. Worldwide, campylobacteriosis is responsible for approximately 5-14 percent of all cases of diarrhea (Institute for International Cooperation in Animal Biologics, Center for Food Security and Public Health, 2005). Several cases of human clinical illness attributable to water contaminated by livestock have been documented (Lind, 1996; Atwill, 1995).

Escherichia Coli 0157: H7: *E. Coli O 157:H7* is a human pathogen that can cause colitis and in some cases hemolytic uremia syndrome. Cattle have been implicated as a main source of contamination in water-borne and food-borne *E.coli 0157-H7* outbreaks and sporadic infections. Complications and deaths are more frequent in young children, the elderly and those with debilitating illnesses. In the United States, approximately 73 000 infections are reported to occur yearly (Institute for International Cooperation in Animal Biologics, Center for Food Security and Public Health, 2004; Renter *et al.*, 2003; Shere *et al.*, 2002; Shere, Bartless and Kasper, 1998).

Salmonella spp.: Livestock are an important source for several *Salmonella spp.* infectious to humans. *Salmonella dublin* is one of the more frequently isolated serotypes from cattle and a serious food-borne pathogen for humans. Surface water contaminated with bovine *S. dublin* or foods rinsed in contaminated water may serve as vehicles of human infection. *Salmonella spp.* have been isolated from 41 percent of turkeys tested in California (United States) and 50 percent of chickens examined in Massachusetts (United States) (Institute for International Cooperation in Animal Biologics, Center for Food Security and Public Health, 2005; Atwill, 1995).

Clostridium botulinum: *C. botulinum* (the organism that causes botulism) produces potent neurotoxins. Its spores are heat-resistant and can survive in foods that are incorrectly or minimally processed. Among the seven serotypes, types A, B, E and F cause human botulism, while types C and D cause most cases of botulism in animals. *C. botulinum* can be transported through runoff from fields (Carney, Carty and Colwell, 1975; Notermans, Dufreme and Oosterom, 1981).

Viral diseases: Several viral diseases can also be of veterinary importance and may be associated with drinking water such as Picornavirus infections (Foot-and-mouth disease, Teschen/Talfan disease, Avian encephalomyelitis, Swine vesicular disease, Encephalomyocarditis); Parvovirus infections; Adenovirus infections; Rinderpest virus; or Swine fever.

Livestock parasitic diseases are transmitted either by ingesting environmentally robust transmissive stages (spores, cysts, oocysts, ova, larval and encysted stages) or via use of contaminated water in food processing or preparation, or via direct contact with infective parasitic stages. Cattle act as a source of parasites for human beings and many wildlife species (Olson *et al.*, 2004; Slifko, Smith and Rose, 2000). Excretion of transmissible stages can be high, and the threat to veterinary public health may extend far beyond the contamination areas (Slifko, Smith and Rose, 2000; Atwill, 1995). Among the parasites the most important water-related public health hazards are *Giardia spp., Cryptosporidia spp., Microsporidia spp.* and *Fasciola spp.*

Giardia lamblia and Cryptosporidium parvum: Both are protozoan microbes that can cause gastrointestinal illness in humans (Buret *et al.*, 1990; Ong, 1996). *G. lamblia* and *C. parvum* have become significant water-borne pathogens as they are indigenous infections in many animal species. Their oocysts are small enough to contaminate groundwater, and *C. parvum* oocysts cannot be successfully removed by common water treatment (Slifko, Smith and Rose, 2000; East Bay Municipal Utility District, 2001; Olson *et al.*, 2004). Worldwide, the prevalence in human population is 1 to 4.5 percent in developed countries and 3 to 20 percent in developing countries (Institute for International Cooperation in Animal Biologics, Center for Food Security and Public Health, 2004).

Microsporidia spp: *Microsporidia spp* are intracellular spore-forming protozoa. Fourteen species are identified as opportunistic or emerging pathogens for human beings. In developing countries, *Microsporidia* species represent an even greater public health hazard, as infections were found predominantly in immuno-compromised individuals. The disease is usually borne, but it is also a potential emerging meat-borne zoonosis, which may also be acquired from raw or lightly cooked fish or crustaceans. The presence of human pathogenic *Microsporidia* in livestock or companion animals has been widely reported. *Enterocytozoon bieneusi* (the most frequently diagnosed species in humans) was reported in pigs, cattle, cats, dogs, llama and chickens (Slifko, Smith and Rose, 2000; Fayer *et al.*, 2002).

Fasciola spp.: Fasciolosis (*Fasciola hepatica* and *Fasciola gigantica*) is an important parasitic infection of herbivores and a food-borne zoonosis. The most common transmission route is the ingestion of contaminated water. Food (such as salads) contaminated with metacercariae-contaminated irrigation water may also be a possible transmission route (Slifko, Smith and Rose, 2000; Conceição *et al.*, 2004; Velusamy, Singh and Raina, 2004).

Drug residues contaminate aquatic environments

Pharmaceuticals are used in large quantities in the livestock sector, mainly antimicrobials and hormones. Antimicrobials have a variety of use. They are given for therapeutic purposes to animals but are also given prophylactically to entire groups of healthy animals, typically during stressing events with high risk of infections, such as after weaning or during transport. They are also given to animals routinely in feed or water over longer periods of time to improve growth rates and feed efficiency. When antimicrobials are added to feed or water at lower-than-therapeutic rates some scientists refer to them as "subtherapeutic" or "nontherapeutic" uses (Morse and Jackson, 2003; Wallinga, 2002). Hormones are used to increase feed conversion efficiency, particularly in the beef and pig sector. Their use is not permitted in a series of countries, particularly in Europe (FAO, 2003a).

In developed countries, drug use for animal production represents a high share of total use. About half of the 22.7 million kg of antibiotics produced in the United States annually is used on animals (Harrison and Lederberg, 1998). The Institute of Medicine (IOM) estimates that about 80 percent of the antibiotics administered to livestock in the United States are used for non-therapeutic reasons, i.e. for disease prophylaxis and growth promotion (Wallinga, 2002). In Europe, the amount of antibiotics used decreased after 1997, as a result of prohibition of some of the substances and public discussion on their use. In 1997, 5 093 tonnes were used, including 1 599 tonnes as growth promoters (mostly polyether antibiotics). In 1999, in EU-15 (plus Switzerland) 4 688 tonnes of antibiotics were used in livestock production systems. Of these 3 902 tonnes (83 percent) were used for therapeutic reasons (tetracyclines were the most common group) while only 786 tonnes were used as growth promoters. The four feed additives substances left in the EU (monensin, avilamycin, flavomycin and salinomycin) will be prohibited in the EU by 2006 (Thorsten *et al.*, 2003). The World Health Organization (WHO) has recently called for a ban on the practice of giving healthy animals antibiotics to improve their productivity (FAO, 2003a).

No data are available on the amounts of hormones used in the different countries. Endocrine disruptors interfere with the normal function of body hormones in controlling growth, metabolism and body functions. They are used in feedlots as ear implants or as feed additives (Miller, 2001). The natural hormones commonly used are: estradiol (estrogen), progesterone, and testosterone. The synthetic ones are: zeranol, melengestrol acetate, and trenbolone acetate. Around 34 countries have approved hormones for use in beef production. Among them are the, Australia, Canada, Chile, Japan, Mexico, New

Zealand, South Africa and the United States. When hormones are used cattle experience an 8 to 25 percent increase in daily weight gain with up to a 15 percent gain in feed efficiency (Canadian Animal Health Institute, 2004). No negative direct impacts on human health as a result of their correct application have been scientifically proven. However, the EU, partly in response to consumer pressure, has taken a strict stand on the use of hormones in livestock production (FAO, 2003a).

However, a substantial portion of the drugs used is not degraded in the animal's body and ends up in the environment. Drug residues including antibiotics and hormones have been identified in various aquatic environments including groundwater, surface water, and tap water (Morse and Jackson, 2003). The US Geological Survey found antimicrobial residues in 48 percent of 139 streams surveyed nationwide and animals were considered possible contributors, especially where manure is spread over agricultural land (Wallinga, 2002). For hormones, Estergreen *et al*. (1977) reported that 50 percent of progesterone administrated to cattle was excreted in the faeces and 2 percent in the urine. Shore *et al*. (1993) found that testosterone was readily leached from soil, but estradiol and estrone were not.

Since even low concentrations of antimicrobials exert a selective pressure in freshwater, bacteria are developing resistance to antibiotics. Resistance can be transmitted through the exchange of genetic material between microorganisms, and from non-pathogenic to pathogenic organisms. Because they can confer an evolutionary advantage, such genes spread readily in the bacterial ecosystem: bacteria that acquire resistance genes can out-compete and propagate faster than non-resistant bacteria (FAO, 2003a; Harrison and Lederberg, 1998; Wallinga, 2002). Beside the potential spread of antibiotic resistances, this represents a source of considerable environmental concern.

With hormones, the environmental concern relates to their potential effects on crops and possible endocrine disruption in humans and wildlife (Miller, 2001). Trenbolone acetate can remain in manure piles for more than 270 days, suggesting that water can be contaminated by hormonally active agents through runoff for example. The links between livestock use of hormones and associated environmental impacts is not easily demonstrated. Nevertheless, it would explain wildlife showing developmental, neurologic, and endocrine alterations, even after the ban of known estrogenic pesticides. This supposition is supported by the increasing number of reported cases of feminization or masculinization of fish and the increased incidence of breast and testicular cancers and alterations of male genital tracts among mammals (Soto *et al*., 2004).

Antimicrobials and hormones are not the only drugs of concern. For example, high quantities of detergents and disinfectants are used in dairy production. Detergents represent the biggest portion of chemicals used in dairy operations. High levels of antiparasitics are also used in livestock production system (Miller, 2002; Tremblay and Wratten, 2002).

Heavy metals use in feed return to the environment

Heavy metals are fed to livestock, at low concentrations, for health reasons or as growth promoters. Metals that are added to livestock rations may include copper, zinc, selenium, cobalt, arsenic, iron and manganese. In the pig industry, copper (Cu) is used to enhance performance as it acts as an antibacterial agent in the gut. Zinc (Zn) is used in weaner-pig diets for the control of post-weaning diarrhoea. In the poultry industry Zn and Cu are required as they are enzyme co-factors. Cadmium and selenium are also used and have been found to promote growth in low doses. Other potential sources of heavy metals in the livestock diet include drinking water, some limestone and the corrosion of metal used in livestock housing (Nicholson, 2003; Miller, 2001; Sustainable Table, 2005).

Animals can absorb only 5 to 15 percent of the metals they ingest. Most of the heavy metals they ingest are, therefore, excreted and return to the environment. Water resources can also be contaminated when foot baths containing Cu and Zn are used as hoof disinfectants for sheep and cattle (Nicholson, 2003; Schultheiß *et al*., 2003; Sustainable Table, 2005).

Heavy metal loads deriving from livestock have been analysed locally. In Switzerland, in 1995, it was found that the total heavy metal load in manures amounted to 94 tonnes of copper, 453 tonnes of zinc, 0.375 tonnes of cadmium and 7.43 tonnes of lead from a herd of 1.64 million cattle and 1.49 million pigs (FAO, 2006b). Of this load 64 percent (of the zinc) to 87 percent (of the lead) were in cattle manure (Menzi and Kessler, 1998). Nevertheless, the highest concentration of copper and zinc was found in pig manure.

Pollution pathways

1. Point-source pollution from intensive production systems

As presented in Chapter 1, the major structural changes occurring in the livestock sector today are associated with the development of industrial and intensive livestock production systems. These systems often involve large numbers of animals concentrated in relatively small areas and in relatively few operations. In the United States for example, 4 percent of the cattle feedlots represent 84 percent of cattle production. Such concentrations of animals creates enormous volumes of waste that have to be managed in order to avoid water contamination (Carpenter *et al*., 1998). The way the waste is managed varies widely and the associated impacts on water resources vary accordingly.

In developed countries regulatory frameworks exist, but rules are often circumvented or violated. For example in the State of Iowa (United States) 6 percent of 307 major manure spills were from deliberate actions such as pumping manure onto the ground or deliberate breaches of storage lagoons, while 24 percent were caused by failure or overflow of a manure storage structure (Osterberg and Wallinga, 2004). In the United Kingdom, the number of reported pollution incidents related to farm wastes increased in Scotland from 310 in 1984 to 539 in 1993, and in England and Northern Ireland from 2 367 in 1981 to 4 141 in 1988. Runoff, from intensive livestock production units, is also one of the major sources of pollution in countries where the livestock sector is intensified.

In developing countries, and in particular in Asia, structural change in the sector, and subsequent changes in manure management practices, have caused similar negative environmental impacts. The growth in scale and geographical concentration in the vicinity of urban areas are causing gross land/livestock imbalances that hamper manure recycling options such as use as fertilizer on cropland. In such conditions, the costs of transporting manure to the field are often prohibitive. In addition, peri-urban land is too expensive for affordable treatment systems such as lagooning. As a result, most of the liquid manure from such operations is directly discharged into waterways. This pollution takes place amid high human population densities, increasing the potential impact on human welfare. Treatment is only practised on a minority of farms and is largely insufficient to reach acceptable discharge standards. Although related regulations are in place in developing

© LEAD/HARALD MENZI

Animal waste lagoon in a pig farm – Central Thailand 2000 countries, they are rarely enforced. Even when waste is collected (such as in a lagoon) a considerable part is often lost by leaching or by overflow during the rainy season, contaminating surface water and groundwater resources (Gerber and Menzi, 2005).

Since most pollution goes unrecorded, there is a lack of data, and so a comprehensive evaluation of the level of livestock-related point-source pollution at the global level is not possible. Looking at the global distribution of intensive livestock production systems (see Map 14 and 15, Annex 1) and based on local studies highlighting the existence of direct water contamination by intensive livestock activities, it is clear that much of the pollution is focussed in areas with high density of intensive livestock activities. These areas are mainly located in the United States (Western and Eastern coasts), in Europe (Western France, Western Spain, England, Germany, Belgium, the Netherlands, Northern Italy, and Ireland), in Japan, China and Southeast Asia (Indonesia, Malaysia, the Philippines, Taiwan Province of China, Thailand, Viet Nam), in Brazil, Ecuador, Mexico, Venezuela and in Saudi Arabia.

2. Non-point source pollution from pastures and arable land

The livestock sector can be linked to three main non-point source mechanisms.

First, part of livestock wastes and, in particular, manure are applied on land as fertilizer for food and feed production.

Second, in extensive livestock production systems surface water contamination by waste may come from direct deposition of faecal material into waterways, or by runoff and subsurface flow when deposited on the soil.

Third, livestock production systems have a high demand for feed and forage resources that often require additional inputs such as pesticides or mineral fertilizers that may contaminate water resources after being applied on land (this aspect will be further described in Section 4.3.4).

Manure is spread onto a field in Wisconsin – United States

Polluting agents deposited on rangelands and agricultural lands may contaminate ground and surface water resources. Nutrients, drug residues, heavy metals or biological contaminants applied on land can leach through the soil layers or can be washed away via run off. The extent to which this happens depends on soil and weather characteristics, the intensity, frequency and period of grazing and the rate at which manure is applied. In dry conditions, overland flow events may not be frequent, so most faecal contamination is the result of an animal defecating directly into a waterway (Melvin, 1995; East Bay Municipal Utility District, 2001; Collins and Rutherford, 2004; Miner, Buckhouse and Moore, 1995; Larsen, 1995; Milchunas and Lauenroth, 1993; Bellows, 2001; Whitmore, 2000; Hooda *et al.*, 2000; Sheldrick *et al.*, 2003; Carpenter *et al.*, 1998).

The degree of land degradation has an effect on the mechanisms and amounts of pollution. As plant cover is reduced, and as soil detachment and subsequent erosion are increased, runoff also increases, and so does the transport of nutrients, biological contaminants, sediments and other contaminants to water courses. The livestock sector has a complex impact, as it represents an indirect and direct source of pollution, and also influences directly (via land degradation) the natural mechanisms that control and mitigate pollution loads.

The application of manure on agricultural lands is motivated by two compatible objectives. First (from an environmental and/or economic viewpoint) it is an effective organic fertilizer and reduces the need for purchased chemical inputs. Second, it usually is a cheaper option than treating manure to meet discharge standards.

Nutrients recovered as manure and applied on agricultural lands were estimated globally at 34 million tonnes of N and 8.8 million tonnes of P in 1996 (Sheldrick, Syers and Lingard, 2003). The contribution of manure to total fertilizers has been declining. Between 1961 and 1995, the relative percentages decreased for N from 60 percent to 30 percent, and for P from 50 percent to 38 percent (Sheldrick, Syers and Lingard, 2003). Nevertheless, for many developing countries manure remains the main nutrient input to agricultural lands (see Table 4.11). The biggest contribution rates of manure to fertilization are observed in Eastern Europe and the CIS (56 percent) and sub-Saharan Africa (49 percent) These high rates, especially in sub-Saharan Africa, reflect abundance of land and the high economic value of manure as fertilizer, compared to mineral fertilizer, which may be unaffordable or not available at all in some places

The use of manure as fertilizer should not be considered as a potential threat to water pollution but more as a means to reduce it. When appropriately used, recycling of livestock manure reduces the need for mineral fertilizer. In countries where the recycling rate and the relative contribution from manure to total N application are low there is obviously a need for better manure management.

Using manure as a source of organic fertilizer presents other advantages regarding water pollution by nutrients. Since a high share of the N contained in manure is present in organic form, it becomes available for crops only gradually. Furthermore, the organic matter contained in the manure improves soil structure, and increases water retention and cation exchange capacity (de Wit *et al.*, 1997). Nevertheless, the organic N is also mineralized at times with low N uptake of crops. At such times the N released is most vulnerable to leaching. In Europe a large part of water contamination by nitrate is the result of the mineralization of organic N in autumn and spring.

When the primary function sought from manure application is as a cost-effective organic fertilizer, its use has traditionally been based on N rather than P uptake by crops. However, as the intake rates of N and P by crops are different from the N/P ratio in livestock excreta, this situation has often resulted in an increased level of P in manured soils over time. As the soil is not an infinite sink for P, this situation resulted in an increasing leaching process for P (Miller, 2001). Furthermore, when manure is used as a soil conditioner the dose of P applied on the land often exceeds the agronomic demand and P levels build up in soils (Bellows, 2001; Gerber and Menzi, 2005).

When the primary function sought from manure application is as a cost-effective waste-management practice, crop farmers tend to apply manure at rates that are excessive in intensity and frequency and may also be mistimed and exceeding the vegetation demand. Over-application is mainly driven by high transport and labour costs, which often limit the use of manure as an organic fertilizer to the direct vicinity of industrialized livestock production systems. As a result, manure is applied in excess, leading to accumulation in the soil and water contamination through runoff or leaching.

Nutrient accumulation in soils is reported worldwide. For example, since in the United States and Europe only 30 percent of the P input in fertilizer is taken up in agricultural produce, it is estimated that there is an average accumulation rate of 22 kg of P/ha/yr (Carpenter *et al.*, 1998). The impact of livestock intensification on nutrient balance was analysed in Asia by Gerber *et al.* (2005), see Box 4.2.

P losses to watercourses are typically estimated to be in the range of 3 to 20 percent of

Table 4.11

Global N and P application on crops and pasture from mineral fertilizer and animal manure

Region/country	Crops				Pasture				Contribution of manure to N fertilization
		Mineral fertilizer	Manure			Mineral fertilizer	Manure		
	Area	N	N	P	Area	N	N	P	
	million ha	*(........... thousand tonnes ....)*			*million ha*	*(........... thousand tonnes ....)*			*Percentage*
North America									
Canada	46.0	1 576.0	207.0	115.3	20.0	0.0	207.0	115.3	22
United States	190.0	11 150.0	1 583.0	881.7	84.0	0.0	1 583.0	881.7	
Central America	40.0	1 424.0	351.0	192.4	22.0	25.0	351.0	192.4	43
South America	111.0	2 283.0	1 052.0	576.8	59.0	12.0	1 051.0	576.2	
North Africa	22.0	1 203.0	36.0	18.5	10.0	0.0	34.0	17.4	10
West Asia	58.0	2 376.0	180.0	92.3	48.0	0.0	137.0	70.2	
Western Africa	75.0	156.0	140.0	71.9	26.0	0.0	148.0	76.0	
Eastern Africa	41.0	109.0	148.0	76.0	24.0	31.0	78.0	40.0	49
Southern Africa	42.0	480.0	79.0	40.6	50.0	3 074.0	3 085.0	1 583.8	
OECD Europe	90.0	6 416.0	3 408.0	1 896.7	18.0	210.0	737.0	410.2	38
Eastern Europe	48.0	1 834.0	757.0	413.4	177.0	760.0	2 389.0	1 304.5	56
Former Soviet Union	230.0	1 870.0	2 392.0	1 306.2	13.0	17.0	167.0	91.2	
South Asia	206.0	12 941.0	3 816.0	1 920.9	10.0	0.0	425.0	213.9	
East Asia	95.0	24 345.0	5 150.0	3 358.3	29.0	0.0	1 404.0	915.5	10
Southeast Asia	87.0	4 216.0	941.0	512.0	15.0	0.0	477.0	259.5	
Oceania	49.0	651.0	63.0	38.9	20.0	175.0	52.0	32.1	29
Japan	4.0	436.0	361.0	223.0	0.0	27.0	59.0	36.4	
World	**1 436.0**	**73 467.0**	**20 664.0**	**11 734.7**	**625.0**	**4 331.0**	**12 384.0**	**6 816.6**	**30**

Note: Data refers to 1995.
Source: FAO/IFA (2001).

the P applied (Carpenter *et al.*, 1998; Hooda *et al.*, 1998). N losses in runoff are usually under 5 percent of the applied rate in the case of fertilizer (see Table 4.12). However, this figure does not reflect the true contamination level, as it does not include infiltration and leaching. In fact, overall N export from agricultural ecosystems to water, as a percentage of fertilizer input, ranges from 10 percent to 40 percent from loam and clay soils to 25 to 80 percent for sandy soils (Carpenter *et al.*, 1998). These estimates are consistent with figures provided by Galloway *et al.* (2004) who estimate that 25 percent of the N applied escapes to contaminate water resources.

Nutrient losses from manured lands and their potential environmental impacts are significant. Based on the above figures, we can estimate that every year 8.3 million tonnes of N and 1.5 million tonnes of P coming from manure end up contaminating freshwater resources. The biggest contributor is Asia with 2 million tonnes of N and 0.7 million tonnes of P (24 percent and 47 percent respectively of global losses from manured lands).

Livestock manure can also contribute significantly to heavy metal loads on crop fields. In England and Wales, Nicholson *et al.* (2003) estimated that approximately 1 900 tonnes of

Table 4.12

Estimated N and P losses to freshwater ecosystems from manured agricultural lands

Region	N from animal manure		N losses to freshwater courses	P from animal manure		P losses to freshwater courses
	Crops	Pasture		Crops	Pasture	
	(.............................. thousand tonnes ....)					
North America						
Canada	207.0	207.0	104.0	115.3	20.0	16.2
United States	1 583.0	1 583.0	792.0	881.7	84.0	115.9
Central America	351.0	351.0	176.0	192.4	22.0	25.7
South America	1 052.0	1 051.0	526.0	576.8	59.0	76.3
North Africa	36.0	34.0	18.0	18.5	10.0	3.4
West Asia	180.0	137.0	79.0	92.3	48.0	16.8
Western Africa	140.0	148.0	72.0	71.9	26.0	11.7
Eastern Africa	148.0	78.0	57.0	76.0	24.0	12.0
Southern Africa	79.0	3 085.0	791.0	40.6	50.0	10.9
OECD Europe	3 408.0	737.0	1 036.0	1 896.7	18.0	229.8
Eastern Europe	757.0	2 389.0	787.0	413.4	177.0	70.8
Former Soviet Union	2 392.0	167.0	640.0	1 306.2	13.0	158.3
South Asia	3 816.0	425.0	1 060.0	1 920.9	10.0	231.7
East Asia	5 150.0	1 404.0	1 639.0	3 358.3	29.0	406.5
Southeast Asia	941.0	477.0	355.0	512.0	15.0	63.2
Oceania	63.0	52.0	29.0	38.9	20.0	7.1
Japan	361.0	59.0	105.0	223.0	0.0	26.8
World	**20 664.0**	**12 384.0**	**8 262.0**	**11 734.7**	**625.0**	**1 483.2**

Source: FAO and IFA (2001); Carpenter *et al.* (1998); Hooda *et al.* (1998); Galloway *et al.* (2004).

zinc (Zn) and 650 tonnes of copper (Cu) were applied to agricultural land in the form of livestock manure in 2000, representing 38 percent of annual Zn input (see Table 4.13). In England and Wales, cattle manure is the biggest contributor to heavy metal deposition by manure, mainly because of the large quantities produced rather than to elevated metal contents (Nicholson *et al.*, 2003). In Switzerland manure is responsible for about two-thirds of the Cu and Zn load in fertilizers and for about 20 percent of the Cd and Pb load (Menzi and Kessler, 1998).

There is growing awareness that the heavy metal content in the soil is increasing in many locations and that critical levels could be reached within the foreseeable future (Menzi and Kessler, 1998; Miller, 2001; Schultheiß *et al.*, 2003).

Within pastures, livestock are an additional source of P and N input to the soil in the form of urine and dung patches. Animals generally do not graze uniformly across a landscape. Nutrient impacts concentrate most where animals congregate, and they vary depending on grazing, watering, travelling and resting behaviours. When not taken up by plants or volatized into the atmosphere, these nutrients may contaminate water resources. Plant capacity to mobilize nutrients is overwhelmed most of the time by the high instantaneous local application rate of nutrients. Indeed, in improved cattle grazing systems, the daily urine excretion per urination of a grazing cow is of the order of 2 litres applied to an area of about 0.4 m^2. This represents an instantaneous application of 400–1 200 kg N per

Table 4.13

Heavy metal inputs to agricultural land in England and Wales in 2000

Source		Inputs per year (tonnes)							
		Zn	Cu	Ni	Pb	Cd	Cr	As	Hg
Atmospheric deposition		2 457	631	178	604	21	863	35	11
Livestock manure		1 858	643	53	48	4.2	36	16	0.3
Sewage sludge		385	271	28	106	1.6	78	2.9	1.1
Industrial waste		45	13	3	3	0.9	3.9	n.d.	0.1
Inorganic fertilizer	Nitrogen	19	13	2	6	1.2	4	1.2	<0.1
	Phosphate	213	30	21	3	10	104	7.2	<0.1
	Potash	3	2	<1	1	0.2	1	0.2	<0.1
	Lime	32	7	15	6	0.9	17	n.d.	n.d.
	Total	266	53	37	16	12	126	8.5	0.1
Agrochemicals		21	8	0	0	0	0	0	0
Irrigation Water		5	2	<1	<1	<0.1	<1	0.1	n.d.
Composts		<1	<1	<1	<1	<0.1	<1	n.d.	<0.1
Total		**5 038**	**1 621**	**299**	**778**	**40**	**327**	**62**	**13**

Note: n.d. - no data.
Source: Nicholson *et al.* (2003).

hectare which exceeds annual grass mobilization capacity of 400 kg N ha^{-1} in temperate climates. These patterns often lead to a redistribution of nutrients across the landscape, generating local point sources of pollution. Furthermore, this high instantaneous application of nutrients may burn the vegetation (high plant root toxicity), impairing the natural recycling process for months (Milchunas, and Lauenroth; Whitmore, 2000; Hooda *et al.*, 2000).

At the global level, 30.4 million tonnes of N and 12 millions tonnes of P are deposited annually by livestock in grazing systems. The direct deposition of manure on pastures is extremely important in Central and South America, which represent 33 percent of the global direct deposition for N and P. Nevertheless, this is greatly underestimated as it only includes pure grazing systems. Mixed systems also contribute to the direct deposition of N and P on grazed fields. This adds to the organic or mineral fertilizers applied on grasslands and poses an additional threat to water quality.

Within pastures the effects of grazing intensity on surface water are varied. Moderate grazing intensity does not usually increase P and N losses in runoff from pasture and, therefore, does not affect water resources significantly (Mosley *et al.*, 1997). However, intensive grazing activities do generally increase P and N losses in runoff from pasture and increase N leaching to groundwater resources (Schepers, Hackes and Francis, 1982; Nelson, Cotsaris and Oades, 1996; Scrimgeour and Kendall, 2002; Hooda *et al.*, 2000).

4.3.2 Wastes from livestock processing

Slaughterhouses, meat-processing plants, dairies and tanneries have a high polluting potential locally. The two polluting mechanisms of concern are the direct discharge of wastewater into freshwater courses, and surface runoff originating from processing areas. Wastewater usually contains high levels of total organic carbon (TOC) resulting in high biological oxygen demand (BOD), which leads to a reduction of oxygen levels in water and suppression of many aquatic species. Polluting compounds also include N, P and chemicals from tanneries including toxic

Box 4.2 Impact of livestock intensification on nutrient balance in Asia

Livestock distributions in Asia have two major patterns. In South Asia and western China, ruminants dominate. In these areas production systems are mixed or extensive, mostly traditional, and livestock densities follow agro-ecological land and climate patterns. In India, ruminants account for more than 94 percent of the excretion of P_2O_5. This preponderance of the contribution of ruminants to P_2O_5 excretion is also noted in Bangladesh, Bhutan, Cambodia, Laos, Myanmar and Nepal, where ruminants contribute to more than 75 percent of the excreted P_2O_5.

On the other hand, East and Southeast Asia are dominated by pigs and poultry. Monogastrics (pigs and poultry) account for more than 75 percent of the excreted phosphorus (P_2O_5) in large parts of China, Indonesia, Malaysia and Viet Nam around urban centres.

There is a strong heterogeneity across the study area regarding the P_2O_5 balance, from areas estimated to have a negative balance (mass balance lower than 10 kg/ha) to areas with high surpluses (mass balance higher than 10 kg/ha). For the whole study area, 39.1 percent of agricultural land is estimated to be in a balanced situation with regard to P_2O_5 (MASS BALANCE - 10 to +10 kg P_2O_5), while 23.6 percent is classified as overloaded - mainly in eastern China, the Ganges basin and around urban centres such as Bangkok, Ho Chi Minh City and Manila, with especially high surpluses at the periphery of urban centres.

On average, livestock manure is estimated to

compounds such as chromium (de Haan, Steinfeld and Blackburn, 1997).

Slaughterhouses

A high potential to pollute locally

In developing countries the lack of refrigerated systems often leads to the siting of abattoirs in residential areas to allow delivery of fresh meat. A wide variety of slaughter sites and levels of technology exist. In principle, large scale industrial processing facilitates a higher utilization of by-products such as blood and facilitates the implementation of wastewater treatment systems and the enforcement of environmental regulations (Schiere and van der Hoek, 2000; LEAD, 1999). However, in practice large-scale abattoirs often import their technology from developed countries without the corresponding rendering and waste treatment facilities. When proper wastewater-management systems are not in place, local abattoirs may represent a high threat to water quality locally.

Direct discharges of wastewater from slaughter houses are commonly reported in developing countries. Wastewater from abattoirs is contaminated with organic compounds including blood, fat, rumen contents and solid waste such as intestines, hair and horns (Schiere and van der Hoek, 2000). Typically 100 kg of paunch manure and 6 kg of fat are produced as waste per tonne of product. The primary pollutant of concern is blood, which has a high BOD (150 000 to 200 000 mg/litre). Polluting characteristics per tonne of liveweight killed are presented in Table 4.14 and are relatively similar between red meat and poultry slaughterhouses (de Haan, Steinfeld and Blackburn, 1997).

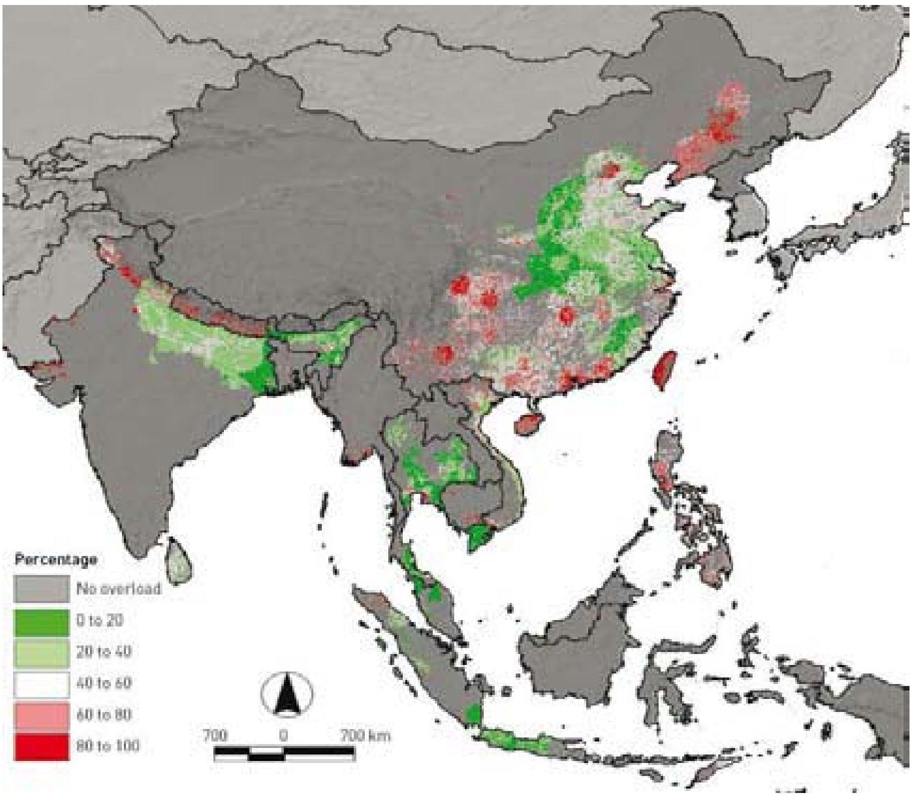
Looking at the European target values for urban waste discharge (e.g. 25 mg BOD, 1 015 mg N and 12 mg P per litre), wastewater from slaughterhouses has a high potential for water pollution even when discharged at low levels. Indeed, if directly discharged into a water course, the wastewater originating from the processing of 1 tonne of read meat contains 5 kg of BOD, which would need to be diluted into 200 000 litres of water in order to comply with EU standards (de Haan, Steinfeld and Blackburn, 1997).

Box 4.2 **cont.**

account for 39.4 percent of the agricultural P_2O_5 supply. Livestock are the dominant agricultural source of P_2O_5 around urban centres and in livestock-specialized areas (southern and northeastern China), while mineral fertilizers are dominant in crop (rice) intensive areas. Mineral fertilizers represent the bulk of the P_2O_5 load in lowlands where rice is the dominant crop: Ganges basin, eastern and southern Thailand, Mekong delta, and eastern China (Jiangsu, Anhui and Henan provinces). On the other hand, manure represents more than half of the phosphate surplus in north-eastern China, southeastern China, Taiwan Province of China, and at the periphery of urban centres such as Hanoi, Ho Chi Minh, Bangkok and Manila.

These observations suggest that there is high potential for better integration of crop and livestock activities. In overloaded areas, part of the mineral fertilizers could, in fact, be substituted by manure, thus substantially decreasing the environmental impacts on land and water. If the potential substitution seems obvious, its implementation on the ground raises a series of issues and constraints (Gerber *et al.*, 2005).

Map 4.1 **Estimated contribution of livestock to total P_2O_5 supply on agricultural land, in an area presenting a P_2O_5 mass balance of more than 10 kg per hectare. Selected Asian countries – 1998 to 2000.**



Source: Gerber *et al.* (2005).

Table 4.14

Typical waste water characteristics from animal processing industries

Operation	BOD	SS	Nkj-N	P
	(........................ kg ....)			
Red meat slaughterhouse (per ton LWK)	5	5.6	0.68	0.05
Red meat packinghouses (per ton LWK)	11	9.6	0.84	0.33
Poultry slaughterhouse (per ton LWK)	6.8	3.5		
Dairies (per ton of milk)	4.2	0.5	<0.1	0.02

Note: LWK – Liveweight killed; SS – Suspended solids; NKj – the Kjeldahl nitrogen is the sum total of organic and ammonia-nitrogen
Source: de Haan, Steinfeld and Blackburn (1997).

Tanneries

Source of wide range of organic and chemical pollutants

The tanning process is a potential source of high local pollution, as tanning operations may produce effluents contaminated with organic and chemical compounds. The individual loads discharged in effluents from individual processing operations are summarized in Table 4.15. Pretanning activities (including cleaning and conditioning hides and skins) produce the biggest share of the effluent load. Water is contaminated with dirt, manure, blood, chemical preservatives and chemicals used to dissolve hairs and epidermis. Acid ammonium salts, enzymes, fungicides, bactericides and organic solvents are widely used to prepare the skins for the tanning process.

Some 80 to 90 percent of the world's tanneries now use chromium (Cr III) salts in their tanning processes. Under conventional modern technologies, 3 to 7 kg of Cr, 137 to 202 kg of Cl^- , 4 to 9 kg of S_2- and 52 to 100 kg of SO_4^{2-} are used per tonne of raw hide. This represents locally a high environmental threat to water resources if adapted wastewater treatments are not in place - as is often the case in developing countries. Indeed in most developing countries tannery effluent is disposed of by sewer, discharged to inland surface water and/or irrigated to land (Gate information services - GTZ, 2002; de Haan, Steinfeld and Blackburn, 1997).

Wastewater from tanneries, with its high concentrations of chromium and hydrogen sulfides, greatly affects local water quality and ecosystems, including fish and other aquatic life. Cr (III) and Cr (VI) salts are known to be carcinogenic compounds (the latter being much more toxic). According to WHO standards, the maximum allowed concentration of chromium for safe drinking water is 0.05mg/l. In areas of high tannery activity the level of chromium in freshwater resources can far exceed this level. When mineral tannery wastewater is applied on agricultural land, soil productivity can be adversely affected, and the chemical compounds used during the tanning process can leach and contaminate groundwater resources (Gate information services GTZ, 2002; de Haan, Steinfeld and Blackburn, 1997; Schiere and van der Hoek, 2000).

Traditional tanning structures (the remaining 10 to 20 percent) use vegetable tanning barks and nuts throughout the entire tanning process. Even if vegetable tannins are biodegradable, they still represent a threat to water quality when used in large quantities. Suspended organic matter (including hair, flesh, and blood residues) originating from the treated skins and vegetable tanning can make water turbid and poses a serious threat to water quality.

Advanced technologies can greatly reduce the pollution loads, especially of chromium, sulphur and ammonia nitrogen (see Table 4.15)

Table 4.15

Pollution loads discharged in effluents from individual tanning operations

		Pollution load (kg/tonnes raw hide)								
Operation	**Technology**	**SS**	**COD**	**BOD**	**Cr**	**S_2-**	**NH_3-N**	**TKN**	**Cl^-**	**SO_4^{2-}**
Soaking	Conventional	11–17	22–33	7–11	–	–	0.1–0.2	1–2	85–113	1–2
	Advanced	11–17	20–25	7–9	–	–	0.1–0.2	0.1–0.2	5–10	1–2
Liming	Conventional	53–97	79–122	28–45	–	3.9–8.7	0.4–0.5	6–8	5–15	1–2
	Advanced	14–26	46–65	16–24	–	0.4–0.7	0.1–0.2	3–4	1–2	1–2
Deliming,	Conventional	8–12	13–20	5–9	–	0.1–0.3	2.6–3.9	3–5	2–4	10–26
Bating	Advanced	8–12	13–20	5–9	–	0–0.1	0.2–0.4	0.6–1.5	1–2	1–2
Tanning	Conventional	5–10	7–11	2–4	2–5	–	0.6–0.9	0.6–0.9	40–60	30–55
	Advanced	1–2	7–11	2–4	0.05–0.1	–	0.1–0.2	0.1–0.2	20–35	10–22
Post-Tanning	Conventional	6–11	24–40	8–15	1–2	–	0.3–0.5	1–2	5–10	10–25
	Advanced	1–2	10–12	3–5	0.1– 0.4	–	0.1–0.2	0.2–0.5	3–6	4–9
Finishing	Conventional	0–2	0–5	2	–	–	–	–	–	–
	Advanced	0–2	0	0	–	–	–	–	–	–
Total	Conventional	83–149	145–231	50–86	3–7	4–9	4–6	12–18	137–202	52–110
	Advanced	35–61	96–133	33–51	0.15–0.5	0.4–0.8	0.6–0.12	5–8	30–55	17–37

Note: COD – chemical oxygen demand; BOD – biological oxygen demand (in five days); SS – suspended solids; TKN – total Kjeldahl nitrogen.
Source: Gate information services – GTZ (2002).

4.3.3 Pollution from feed and fodder production

Over the two last centuries, the increased pressure on agricultural land, associated with poor land management practices, has resulted in increased erosion rates and decreased soil fertility over wide areas. As shown in Chapter 2, the livestock sector has contributed extensively to this process.

Feed production is estimated to account for 33 percent of agricultural crop land (Chapter 2). The increasing demand for food and feed products, combined with declining natural fertility of agricultural lands resulting from increased erosion, led to an increased use of chemical and organic inputs (including fertilizers and pesticides) to maintain high agricultural yields. This increase, in turn, contributed to the widespread pollution of freshwater resources. As we shall see in this section, in most geographical areas the livestock sector should be considered as the major driver for the trend of increasing water pollution.

1. Nutrients

We have already seen (in Section 4.3.1) that manure applied to crops (including feedcrops) can be associated with water pollution. In this section we focus on the fertilization of feedcrops with mineral fertilizers. While the two practices are complementary and are often combined, we have separated them here for clarity of the analysis. Their integration, and the concept of nutrient management plans, will be discussed in the mitigation option section.

The use of mineral fertilizer for feed and food production has increased significantly since the 1950s. Between 1961 and 1980 nitrogenous fertilizer consumption was multiplied by 2.8 (from 3.5 to 9.9 million tonnes per year) and 3.5 (from 3.0 to 10.8 million tonnes per year) in Europe (15) and the United States respectively. Identically the consumption of phosphate fertilizers was multiplied by 1.5 (from 3.8 to 5.7 million tonnes per year) and 1.9 (from 2.5 to 4.9 million tonnes per year) in these regions. Currently, humans release as much N and P to terrestrial ecosystems annually as all natural sources combined. Between 1980 and 2000, global N consumption increased by 33 percent and P consumption by 38 percent. Tilman *et al*. (2001) projected that if past trends in N and P fertilization and irrigation, and their correlation with increasing population and GDP continue, the global N fertilization level would be 1.6 times greater than in 2000 by 2020 and 2.7 times greater than in 2000 by 2050, while P fertilization would be 1.4 times greater by 2020 and 2.4 times greater by 2050.

Changes, at the regional level, show considerable diversity over the last two decades (Table 4.16). Between 1980 and 2000, the increases in the use of mineral fertilizer has been particularly strong in Asia (+117 percent for N and +154 percent for P), Latin America (+80 percent for N and +334 percent for P), and Oceania (+337 percent for N and +38 percent for P). In developed countries there is currently a stagnation (+2 percent for N use in North America) or an actual decline in the use of mineral fertilizer (-8 percent for N and 46 percent for P use in Europe, -20 percent for P use in Northern America). These trends can be explained by the fact that market prices of arable crops have fallen, creating economic pressure for a more accurate matching of fertilizer application rates to crop needs. Furthermore in some areas (Europe for example), owing to environmental concerns, standards and policies have been developed to control application rates, methods and timing. However as most modern crop varieties require relatively high rates of fertilizer application, fertilizer use remains high (Tilman *et al*., 2001; Stoate *et al*., 2001).

Asia is the leading user of mineral fertilizer with 57 percent and 54.5 percent of the global consumption for N and P respectively. In contrast, the consumption of fertilizer in sub-Saharan Africa is still insignificant representing 0.8 percent and 1.2 percent of the global consumption for N and P respectively.

The increased consumption of fertilizer over the past 50 years has made agriculture an ever-increasing source of water pollution (Ongley, 1996; Carpenter, 1998).

The livestock sector is a major cause of this increase. Table 4.17 describes the livestock contribution to N and P consumption in 12 major countries, covering both livestock and feed production. In five of them, livestock are directly or indirectly responsible for more than 50 percent of the mineral N and P applied on agricultural land (i.e. Canada, France, Germany, the United Kingdom and the United States). The extreme case is the United Kingdom, where livestock contributes to 70 percent and 58 percent respectively of the amount of N and P applied on agricultural

Table 4.16

Mineral fertilizer consumption in different world regions between 1980 and 2000

Regions	Nitrogenous fertilizers consumption (tonnes)		Percentage change	Phosphate fertilizers consumption (tonnes)		Percentage change
	1980	2000	1980–2000	1980	2000	1980–2000
Asia	21 540 789	46 723 317	117	6 971 541	17 703 104	154
Commonwealth of Independent States		2 404 253			544 600	
Africa South of Sahara	528 785	629 588	19	260 942	389 966	49
European Union (15)	9 993 725	9 164 633	-8	5 679 528	3 042 459	-46
Latin America and the Caribbean	2 864 376	5 166 758	80	2 777 048	3 701 328	33
Central America	1 102 608	1 751 190	59	325 176	443 138	36
North America	11 754 950	12 028 513	2	5 565 165	4 432 567	-20
Oceania	273 253	1 192 868	337	1 139 807	1 571 016	38
World	**60 775 733**	**80 948 730**	**33**	**31 699 556**	**32 471 855**	**2**

Source: FAO (2006b).

lands. In the four European countries we can also note the high fertilizer rates for pastures. In the United Kingdom for example pasture represents 45.8 percent of N and 31.2 percent of P consumption for agriculture. In these countries we can reasonably surmise that the livestock sector is the leading contributor to water pollution deriving from mineral fertilizers on agricultural lands. In the other countries studied this contribution is also extremely important. In Brazil and Spain, the livestock contribution to agricultural N and P use is over 40 percent. Livestock's contribution is relatively less important in Asia with 16 percent for N use in China and 3 percent for P and N use in India. Nevertheless, even if low in relative value, the volume of N and P used by the livestock sector is extremely high in absolute terms as Asia represents almost 60 percent of the global consumption of N and P mineral fertilizer.

When applied on agricultural lands, nitrogen and phosphate reach watercourses during leaching, surface runoff, subsurface flow and soil erosion (Stoate *et al.*, 2001). The transport of N and P depends on the time and rate of fertilizer application together with land-use management and site characteristics (soil texture and profile, slope, vegetation cover) and climate (rainfall characteristics). The latter particularly influences the leaching process (especially for N) and the contamination of groundwater resources (Singh and Sekhon, 1979; Hooda *et al.*, 2000).

In Europe, NO_3 concentration exceeded the international standards (NO_3:45 mg/litre; NO_3-N:10 mg/litre) in the groundwater below 22 percent of the cultivated land (Jalali, 2005; Laegreid *et al.*, 1999). In the United States an estimated 4.5 million people drink water from wells containing nitrates above the standards (Osterberg and Wallinga, 2004; Bellows, 2001; Hooda *et al.*, 2000). In developing countries numerous assessments have shown the link between high fertilization rates, irrigation, and groundwater pollution by nitrates (Costa *et al.*, 2002; Jalali, 2005; Zhang *et al.*, 1996).

Table 4.17

Contribution of livestock production to agricultural N and P consumption in the form of mineral fertilizer in selected countries

Countries	N (mineral fertilizer) Consumption (thousand tonnes)					P_2O_5 (mineral fertilizer) Consumption (thousand tonnes)				
	Total use for agriculture	Use for feed production	Use for pastures and fodder	Total use	Livestock contribution (%)	Total use for agriculture	Use for feed production	Use for pastures and fodder	Total use	Livestock contribution (%)
Argentina	436.1	126.5	Negligible	126.5	29	336.3	133.7	Negligible	133.7	40
Brazil	1 689.2	678.1	Negligible	678.1	40	1 923.8	876.4	Negligible	876.4	46
China	18 804.7	2 998.6	Negligible	2 998.6	16	8 146.6	1 033.8	Negligible	1 033.8	13
India	10 901.9	286.0	Negligible	286.0	3	3 913.6	112.9	Negligible	112.9	3
Mexico	1 341.0	261.1	1.6	262.7	20	418.9	73.8	0.6	74.4	18
Turkey	1 495.6	243.1	18.6	261.7	17	637.9	108.2	8.0	116.2	18
USA	9 231.3	4 696.9	Negligible	4 696.9	51	4 088.1	2 107.5	Negligible	2 107.5	52
Canada	1 642.7	894.4	3.0	897.4	55	619.1	317.6	1.0	318.6	51
France	2 544.0	923.2	393.9	1 317.1	52	963.0	354.5	145.4	499.9	52
Germany	1 999.0	690.2	557.0	1 247.2	62	417.0	159.7	51.0	210.7	51
Spain	1 161.0	463.3	28.0	491.3	42	611.0	255.0	30.0	285.0	47
United Kingdom	1 261.0	309.2	578.0	887.2	70	317.0	84.3	99.0	183.3	58

Note: Based on 2001 consumption data.
Source: FAO (2006b).

Table 4.18

Estimated N and P losses to freshwater ecosystems from mineral fertilizers consumed for feed and forage production

	N (mineral fertilizer) consumption for feed and forage production	N losses to freshwater ecosystems	P (mineral fertilizer) consumption for feed and forage production	P losses to freshwater ecosystems
	(.....	..... thousand tonnes	.....	.....)
Argentina	126.5	32	133.7	17
Brazil	678.1	170	876.4	105
China	2998.6	750	1033.8	124
India	286	72	112.9	13
Mexico	262.7	66	74.4	9
Turkey	261.7	65	116.2	14
USA	4696.9	1174	2107.5	253
Canada	897.4	224	318.6	38
France	1317.1	329	499.9	60
Germany	1247.2	312	210.7	25
Spain	491.3	123	285	34
United Kingdom	887.2	222	183.3	22

Note: Based on 2001 consumption data.
Source: FAO (2006b); Carpenter *et al.* (1998); Hooda *et al.* (1998) and Galloway *et al.* (2004).

N and P loss rates estimated by Carpenter *et al.* (1998) and Galloway *et al.* (2004) (see Section 4.3.1), were used to estimate N and P losses to freshwater ecosystems from mineral fertilizers consumed for feed and forage production (see Table 4.18). High losses occur especially in the United States (with 1 174 000 tonnes for N and 253 000 tonnes for P), China (750 000 tonnes for N and 124 000 tonnes for P) and Europe.

Accurate estimation of the relative contribution of the livestock sector to N and P water pollution at global level is not possible because of lack of data. However, this relative contribution can be investigated in the United States based on the work presented by Carpenter *et al.*, 1998 (see Table 4.19). Livestock's contribution, including N and P losses from cropland used for feed, pastures and rangelands, represent one-third of total discharge to surface water for both N and P.

We can assume that the livestock sector is probably the leading contributor to water pollution by N and P in the United States.

These impacts represent a cost to society which may (depending on the opportunity value of the resources affected) be enormous. The livestock sector is the first contributor to these costs in several countries. For the United Kingdom, the cost of removing nitrates from drinking-water costs is estimated at US$10 per kg, totalling US$29.8 million per year (Pretty *et al.*, 2000). The costs associated with erosion and P pollution were even higher and were estimated at US$96.8 million. These figures are probably underestimates, as they do not include the costs associated with the impacts on ecosystems.

2. Pesticides used for feed production

Modern agriculture relies on the use of pesticides[7] to maintain high yields. Pesticide use has declined in many OECD countries but is still

[7] Pesticide is a generic term to describe a chemical substance used to kill, control, repel, or mitigate any disease or pest. It includes herbicides, insecticides, fungicides, nematocides and rodenticides (Margni *et al.*, 2002; Ongley, 1996).

Table 4.19

Livestock contribution to nitrogen and phosphorus discharges to surface waters from non-point source and point source pollution in the United States

Source	Total N	Total P	Livestock contribution N losses	Livestock contribution P losses
	(......... Thousand tonnes per year ....)			
Croplands	3 204	615	1 634	320
Pastures	292	95	292	95
Rangelands	778	242	778	242
Forests	1 035	495		
Other rural lands	659	170		
Other non-point sources	695	68		
Other point sources	1 495	330		
Total	158	2015		
Livestock contribution			**2 704**	**657**
Percentage of the total			**33.1**	**32.6**

Source: Based on Carpenter *et al.* (1998).

Spraying pesticide on crops – United States

on the increase in most developing countries (Stoate *et al.*, 2001; Margni *et al.*, 2002; Ongley 1996). Pesticides applied on agricultural land can contaminate the environment (soil, water and air) and affect non-target living organisms and micro-organism, thus damaging the proper functioning of ecosystems. They also constitute a risk to human health through residues in water and in food (Margni *et al.*, 2002; Ongley, 1996).

Several hundred different pesticides are currently used for agricultural purposes around the world. The two most important classes are organochlorine and organophosphorous compounds (Golfinopoulos *et al.*, 2003). Pesticide contamination of surface water resources is reported worldwide. While it is difficult to separate the role of pesticides from that of industrial compounds that are released into the environment, there is evidence that agricultural use of pesticides represents a major threat to water quality (Ongley, 1996). In the United States for example the Environmental Protection Agency's National Pesticide Survey found that 10.4 percent of community wells and 4.2 percent of rural wells contained detectable levels of one or several pesticides (Ongley, 1996).

The main form of loss of pesticides from treated crops is volatilization, but runoff, drainage and leaching may lead to indirect contamination of surface and groundwaters. Direct contamination of water resources may arise during the application of pesticides as they can partly move by air to non-target areas downwind, where they can affect fauna, flora and humans (Siebers, Binner and Wittich, 2003; Cerejeira *et al.*, 2003; Ongley, 1996).

The persistence of pesticides in soils also varies depending on runoff, volatilization and leaching processes and the degradation processes, which vary depending on the chemical stability of the compounds (Dalla Villa *et al.*, 2006). Many pesticides (in particular organophosphorous pesticides) dissipate rapidly in soils as a result of mineralization. But others (organochlorine pesticides) are very resistant and remain active longer in the ecosystem. As they resist biodegradation, they can be recycled through food chains and reach higher concentrations at the top levels of the food chain (Golfinopoulos *et al.*, 2003; Ongley, 1996; Dalla Villa *et al.*, 2006).

Surface water contamination may have ecotoxicological effects on aquatic flora and fauna, and for human health if the water is used for public consumption. The impacts are the outcome of two distinct mechanisms: bioconcentra-

Box 4.3 Pesticide use for feed production in the United States

Agriculture is a major user of pesticides in the United States, accounting for 70 to 80 percent of total pesticide use (United States Geological Survey - USGA, 2003). Herbicides constitute the largest pesticide category in the US agriculture while insecticides are generally applied more selectively and at lower rates.

Soybean and corn are the two most extensively grown field crops, totaling about 62 million hectares in 2005 (FAO, 2006). Corn is the largest herbicide user (USDA-ERA, 2002). In 2001, about 98 percent of the 28 million hectares of corn planted in the major producing states were treated with a total of about 70 000 tonnes of herbicides. However, only 30 percent of the planted corn acreage was treated with insecticides, amounting to about 4 000 tonnes. Soybean production in the US also utilizes significant amounts of herbicides. An estimated 22 000 tonnes of herbicides were applied to 21 million hectares of soybean in 2001 (USDA/NASS, 2001).

Overall pesticide use intensity (defined as the average amount of chemical applied per hectare of planted area) in corn and soy production has declined over the years a decline that can be attributed to technological improvements, the introduction of genetically modified crops, and the increase of pesticide toxicity (reduced application rate) (Ackerman *et al.*, 2003). Nevertheless, owing to the increased toxicity of the compounds used the ecological impacts may not have declined.

In 2001, feed production in the United States

tion and biomagnification (Ongley, 1996). Bioconcentration refers to the mechanisms by which pesticides concentrate in fat tissue over the life of an individual. Biomagnification refers to the mechanisms by which pesticide concentrations increase through the food chain, resulting in high concentration in top predators and humans. Pesticides impact the health of wild animals (including fishes, shellfishes, birds and mammalians) and plants. They can cause cancers, tumours and lesions, disruption of immune and endocrine systems, modification of reproductive behaviours and birth defects (Ongley, 1996; Cerejeira *et al.*, 2003). As a result, of these impacts the whole food chain may be affected.

The contribution of the livestock sector to pesticide use is illustrated for the United States in Box 4.3. In 2001, the volume of herbicide used for US corn and soybean amounted to 74 600 tonnes, 70 percent of the total herbicide use in agriculture. For insecticides the relative contribution of corn and soybean production for feed to total agricultural use declined from 26.3 percent to 7.3 percent between 1991 and 2001, as a result of technological improvements, the introduction of genetically modified crops and the improved toxicity of pesticides (Ackerman *et al.*, 2003). Although the relative contribution of feed production (in the form of soybean and corn) toward pesticide use is declining in the United States (from 47 percent in 1991 to 37 percent in 2001), livestock production systems remain a major contributor to their use.

We can assume that the role of livestock production systems in pesticide use is equally important in other main feed producing countries, including Argentina, Brazil, China, India and Paraguay.

3. Sediments and increased turbidity levels from livestock-induced erosion

Soil erosion is the result of biotic factors, such as livestock or human activity and abiotic, such as wind and water (Jayasuriya, 2003). Soil erosion is a natural process and is not a problem where soil regeneration equals or exceeds soil loss. However, in most parts of the world this is not the case. Soil erosion has increased dramatically because of human activities. Large parts of the world including Europe, India, East and South

Box 4.3 **cont.**

was constituted by corn (43.6 percent), soybean (33.8 percent), wheat (8.6 percent), and sorghum (5.5 percent), the rest being comprised of other oilseeds and grains. In 2001 60 percent of US corn production and 40 percent of soybean production was used utilized for feed (FAO, 2006b). Total quantities of herbicide use for corn and soybean, use intensities, and the herbicide usage by the livestock sector are shown in the table below. Livestock sector usage declined by 20 percent between 1991 and 2001. In 2001, 70 percent of the volume of herbicides used in agriculture can be attributed to animal feed production in the form of soybean and corn. The use of insecticide in corn production for feed declined more strongly over this same period, from 8 200 tonnes (26 percent of total insecticide use in agriculture) to 3,400 tonnes (7 percent). Although the relative contribution of feed (soybean and corn) toward pesticide use is declining in the United States (from 47 percent in 1991 to 37 percent in 2001), livestock production systems still remain a major contributor to their use. Although it may not be possible isolate these impacts on water resources or to draw conclusions on their magnitude, the use of pesticides for feed grain and oilseed production in the United States undoubtedly has major environmental impacts on water quality as well as on water-related ecosystems.

Table 4.20

Pesticide use for feed production in the United States

	1991	1996	2001
Total agricultural herbicide use *(tonnes)*	139 939	130 847	106 765
Total agricultural insecticide use *(tonnes)*	32 185	16 280	51 038
Herbicide use for corn - 100% of the planted area is treated			
Herbicide application rate *(kg/ha)*	3.1	3	2.5
Total herbicide used for feed production *(tonnes)*	70 431	71 299	55 699
Herbicides use in feed production as % of total agricultural herbicide use *(%)*	50.3	54.5	52.2
Insecticide use for corn - 30% of the planted area is treated			
Insecticide application rate *(kg/ha)*	1.2	0.8	0.5
Total insecticides used for feed production *(tonnes)*	8 253	5 781	3 380
Insecticide use in feed production as % of total agricultural insecticide use *(%)*	26	36	7
Herbicide use for soybean - 100 % of the planted area is treated			
Herbicide application rate *(kg/ha)*	1.3	1.3	1.1
Total herbicide used for feed production *(tonnes)*	18 591	19 496	18 882
Herbicide use in feed production (soybean) as a % of total agricultural herbicide use *(%)*	13.3	14.9	17.7
Insecticide use for soybean - 2% of the planted area is treated			
Insecticide application rate *(kg/ha)*	0.4	0.3	0.3
Total insecticides used for feed production *(tonnes)*	108	88	91
Insecticide use in feed production as % of total agricultural insecticide use *(%)*	0.3	0.5	0.3
Total agricultural pesticide use *(tonnes)*	207 382	199 991	211 148
Total Pesticide used for feed production (soybean and corn) as a % of total agricultural pesticide use *(%)*	47	48	37

Source: FAO (2006b); USDA/NASS (2001); USDA-ERA (2002).

China, Southeast Asia, the eastern United States and Sahelian Africa are particularly at risk from human-induced water erosion (see Map 4.2).

Apart from loss of soil and soil fertility, erosion also results in sediments being transported to waterways. Sediments are considered as the principal non-point source water pollutant related to agricultural practices (Jayasuriya, 2003). As a result of erosion processes, 25 billion tonnes of sediments are transported through rivers every year. With the worldwide increased demand for feed and food products the environmental and economic costs of erosion are increasing dramatically.

As presented in Chapter 2, the livestock sector is one of the major contributors to the soil erosion process. Livestock production contributes to soil erosion and, therefore, sediment pollution of waterways in two different ways:

- indirectly, at feed production level when cropland is inappropriately managed or as result of land conversion; and
- directly, through livestock hoof and grazing impacts on pastures.

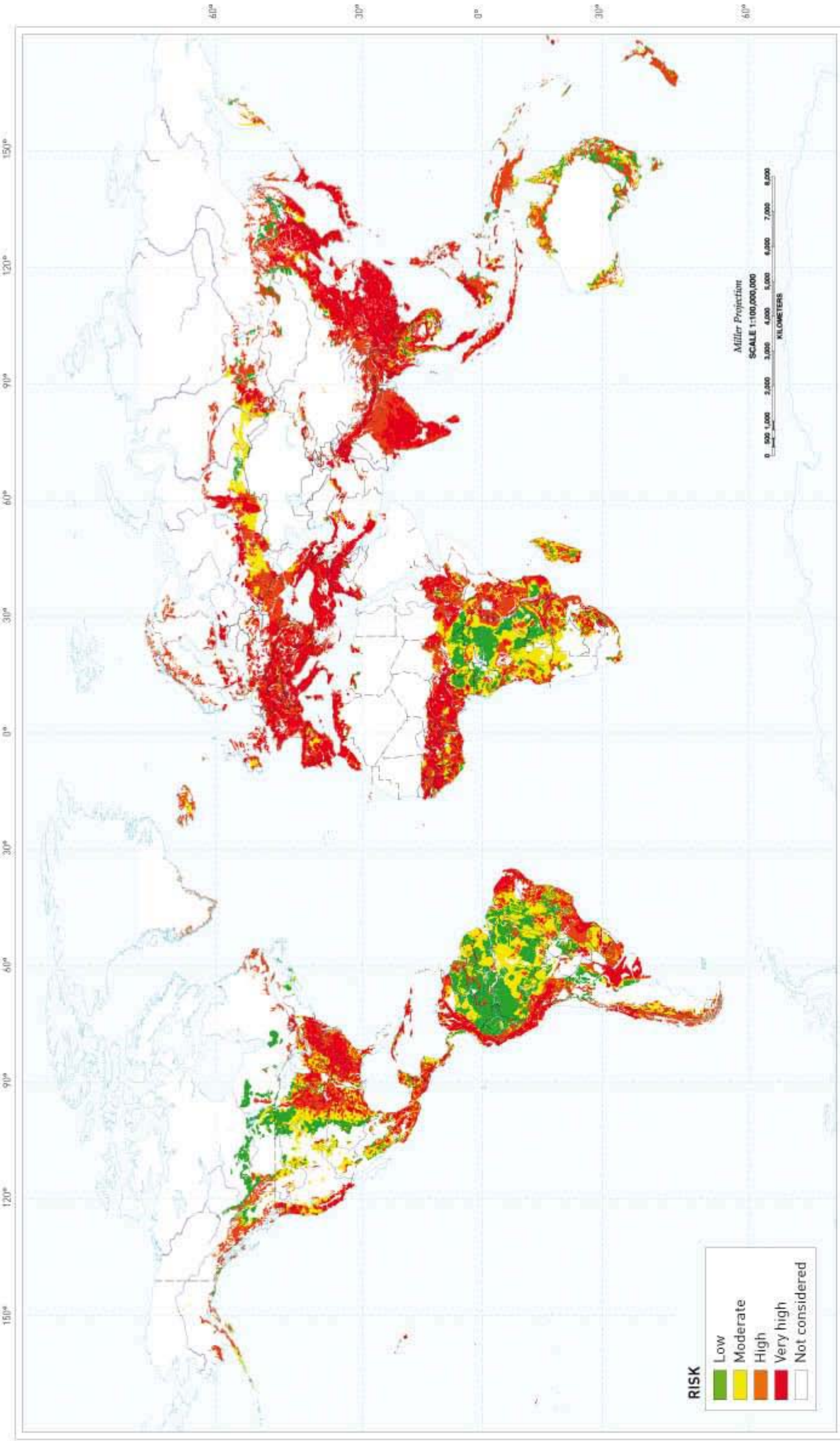
Croplands, especially under intensive agriculture, are generally more prone to erosion than other land uses. Major factors that contribute to increased erosion rates within croplands are developed in Chapter 2. The European Union Environmental Directorate estimates that the mean annual soil loss across northern Europe is higher than 8 tonnes/ha. In Southern Europe 30 to 40 tonnes/ha^{-1} can be lost in a single storm (De la Rosa *et al.*, 2000 cited by Stoate *et al.*, 2001). In the United States about 90 percent of cropland is currently losing soil, above the sustainable rate, and agriculture is identified as the leading cause of impairment of water resources by sediments (Uri and Lewis, 1998). Soil erosion rates in Asia, Africa and South America are estimated to be about twice as high as in the United States (National Park Service, 2004). Not all the eroded top soil goes on to contaminate water resources. Some 60 percent or more of the eroded soil settles out of the runoff before it reaches a water body, and may enhance soil fertility locally, downhill from the areas that are losing soil (Jayasuriya, 2003).

On the other hand, concentrated "hoof action" by livestock in areas such as stream banks, trails, watering points, salting and feeding sites causes compaction of wet soils (whether vegetated or exposed), and mechanically disrupts dry and exposed soils. Compacted and/or impermeable soils can have decreased infiltration rates, and therefore increased volume and velocity of runoff. Soils loosened by livestock during the dry season are a source of sediments at the beginning of the new rainy season. In riparian areas the destabilization of streambanks by livestock activities contributes locally to a high discharge of eroded material. Furthermore, livestock can overgraze vegetation, disrupting its role of trapping and stabilizing soil and aggravating erosion and pollution (Mwendera and Saleem, 1997; Sundquist, 2003; Redmon, 1999; Engels, 2001; Folliott, 2001; Bellows, 2001; Mosley *et al.*, 1997; Clark Conservation District, 2004; East Bay Municipal Utility District, 2001).

The erosion process decreases the on-site water-holding capacity of the soil. The off-site impacts relate to the impairment of water resources and include:

River bank soils loosened by water buffaloes in Naning, China causing sedimentation and turbidity

Map 4.2 Risk of human-induced water erosion



Source: USDA-NRCS (1999).

- Increased sedimentation in reservoirs, rivers and channels resulting in the obstruction of waterways, clogging of drainage and irrigation systems.
- Destruction of aquatic ecosystem habitats. Streambeds and coral reefs are blanketed with fine sediments, which cover food sources and nesting sites. Increased water turbidity reduces the amount of light available in the water column for plant and algae growth, raises surface temperature, affects respiration and digestion among aquatic organisms and covers.
- Disruption of the hydraulic characteristics of the channel, resulting in higher peak flow leading to loss of infrastructure and lives during flooding and reduced water availability during the dry season.
- Transport of adsorbed agricultural nutrients and pollutants, especially phosphorus, chlorinated pesticides and most metals, to reservoirs and watercourses resulting in an accelerated pollution process. The adsorption of sediment is influenced by the size of the particles and the amount of particulate organic carbon associated with the sediment.
- Influence on micro-organisms. Sediments promote growth of micro-organisms and protect them from disinfection processes.
- Eutrophication. The decreased oxygen levels (as a final result of the impairment of ecosystems functioning) may also enhance the development of anaerobic microflora (Ongley, 1996; Jayasuriya, 2003; Uri and Lewis, 1998).

The role of livestock production systems in erosion and increased turbidity levels is illustrated by a United States case study (see Box 2.4, Chapter 2), which identified livestock production systems as the major contributor to soil erosion and its associated water pollution, accounting for 55 percent of the total soil mass eroded from agricultural lands every year. At the global level, we can assume that the livestock production system plays a major role regarding water contamination by sediments in countries with important feed production or with large areas dedicated to pasture.

Increased erosion has economic costs both on-site and off-site. On-site, the loss of top soil represents an economic loss to agriculture through loss of productive land, top soil, nutrients and organic matter. Farmers have to maintain field productivity by using fertilizers that represent a considerable cost and may further pollute water resources. However, many small-scale farmers in developing countries cannot afford to buy these inputs and, therefore, suffer declining yields (Ongley, 1996; Jayasuriya, 2003; UNEP, 2003). Off-site, suspended solids impose costs on water treatment facilities for their removal. Mud removal from stream channels constitutes a considerable cost to local populations. The cost of erosion in the United States in 1997 has been estimated at US$29.7 billion, representing 0.4 percent of GDP (Uri and Lewis, 1998). The costs associated to the increased frequency of flooding events are also massive.

4.4 Livestock land-use impacts on the water cycle

The livestock sector not only contributes to the use and pollution of freshwater resources but also impacts directly the water replenishment process. Livestock's land-use affects the water cycle by influencing water infiltration and retention. This impact depends on the type of land use, and therefore varies with land-use changes.

4.4.1 Extensive grazing alters water flows

Globally 69.5 percent of the rangelands (5.2 billion ha) in dry lands are considered as degraded. Rangeland degradation is widely reported in southern and Central Europe, Central Asia, sub-Saharan Africa, South America, the United States and Australia (see Chapter 2). Half of the 9 million hectares of pasture in Central America are estimated to be degraded, while over 70 percent of the pastures in the northern Atlantic zone of Costa Rica are in an advanced stage of degradation.

Land degradation by livestock has an impact on the replenishment of water resources. Overgrazing and soil trampling can severely compromise the water cycle functions of grasslands and riparian areas by affecting water infiltration and retention, and stream morphology.

Uplands, as the headwaters of major drainage systems that extend to lowlands and riparian areas,[8] make up the largest part of watersheds and play a key role in water quantity and water delivery. In a properly functioning watershed, most precipitation is absorbed by soil in the uplands, and is then redistributed throughout the watershed by underground movement and controlled surface runoff. Any activities that affect the hydrology of the uplands, therefore, have significant impacts on water resources of lowlands and riparian areas (Mwendera and Saleem, 1997; British Columbia Ministry of Forests, 1997; Grazing and Pasture Technology Program, 1997).

Riparian ecosystems increase water storage and groundwater recharge. Soils in riparian areas differ from upland areas, as they are rich in nutrients and organic matter, which allow the soil to retain large amounts of moisture. The presence of vegetation slows down the rain and allows water to soak into the soil, facilitating infiltration and percolation and recharging groundwater. Water moves downhill through the subsoil and seeps into the channel throughout the year, helping to transform what would otherwise be intermittent streams into perennial flows, and extending water availability during the dry season (Schultz, Isenhart and Colletti, 1994; Patten *et al.*, 1995; English, Wilson and Pinkerton,1999; Belsky, Matzke and Uselman, 1999). The vegetation filters out sediment and builds up and reinforces the stability of stream banks. It also reduces the sedimentation of waterways and reservoirs, thereby also increasing water availability (McKergow *et al.*, 2003).

Infiltration separates water into two major hydrologic components: surface runoff and subsurface recharge. The infiltration process influences the source, timing, volume and peak rate of runoff. When precipitation is able to enter the soil surface at appropriate rates, the soil is protected against accelerated erosion and soil fertility can be maintained. When it cannot infiltrate, it runs off as surface flow. Overland flow may travel down slope to be infiltrated on another portion of the hill slope, or it may continue on and enter a stream channel. Any mechanism that affects the infiltration process in the uplands, therefore, has consequences far beyond the local area (Bureau of Land Management, 2005; Pidwirny M., 1999; Diamond and Shanley, 1998; Ward, 2004; Tate, 1995; Harris *et al.*, 2005).

The direct impact of livestock on the infiltration process varies, depending on the intensity, frequency and duration of grazing. In grassland ecosystems, infiltration capacity is mainly influenced by soil structure and vegetation density and composition. When vegetation cover declines, soil organic matter content and aggregate soil stability decrease, reducing the soil's infiltration capacity. Vegetation further influences the infiltration process by protecting the ground from raindrops, while its roots improve soil stability and porosity. When soil layers are compacted by trampling, porosity is reduced and the level of infiltration is reduced dramatically. Thus, when not appropriately managed, grazing activities modify the physical and hydraulic properties of soils and ecosystems, resulting in increased runoff, increased erosion, increased frequency of peak flow events, increased water velocity, reduced late season flow and lowered water tables (Belsky, Matzke and Uselman, 1999; Mwendera and Saleem, 1997).

Generally, grazing intensity is recognized as the most critical factor. Moderate or light graz-

[8] Riparian ecosystems are wetlands adjacent to rivers and lakes, where soils and vegetation are influenced by elevated water tables. In headwater or ephemeral streams, riparian zones are often narrow strips of adjacent land. In large rivers they can be well-developed floodplains. Riparian areas usually result in a combination of high biodiversity, high density of species and high productivity (Carlyle and Hill, 2001; Mosley *et al.*, 1997; McKergow *et al.*, 2003).

ing reduces infiltration capacity to about three-quarters of the un-grazed condition, while heavy grazing reduces infiltration capacity to about half (Gifford and Hawkins, 1978 cited by Trimble and Mendel, 1995). Indeed, livestock grazing influences vegetation composition and productivity. Under heavy grazing pressure, plants may not be able to compensate sufficiently for the phytomass removed by grazing animals. With decreased soil organic matter content, soil fertility and soil aggregate stability, the natural infiltration level is impacted (Douglas and Crawford, 1998; Engels, 2001). Grazing pressure increases the amounts of less desirable vegetation (brush, weedy trees) that may extract water from the deeper soil profile. The changed plant species composition may not be as effective in intercepting raindrops and retarding runoff (Trimble and Mendel, 1995; Tadesse and Peden, 2003; Integrated Resource Management, 2004; Redmon, 1999; Harper, George and Tate, 1996). The period of grazing is also important as when soils are wet they can more easily be compacted and the stream banks can easily be destabilized and destroyed.

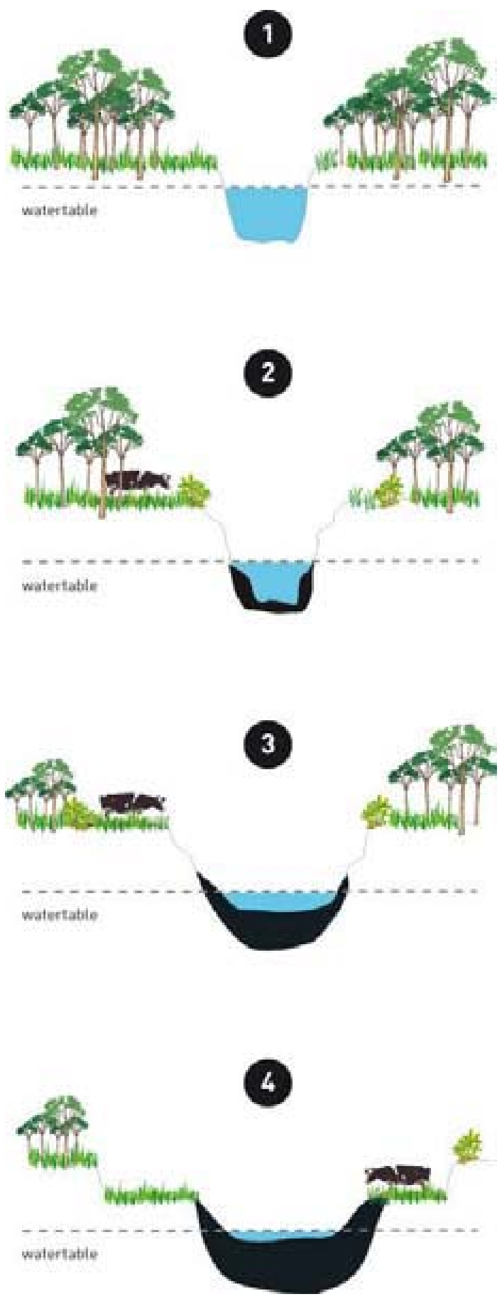
Grazing animals are also important agents of geomorphological change as their hooves physically reshape the land. In the case of cattle, the force is usually calculated as the mass of the cow (500 kg approx.) divided by the basal hoof area (10 cm^2). However, this approach may lead to underestimates, as moving animals may have one or more feet off the ground and the mass is often concentrated on the down slope rear leg. On point locations, cattle, sheep and goats can easily exert as much downward pressure on soil as a tractor (Trimble and Mendel, 1995; Sharrow, 2003).

The formation of compacted layers within the soil decreases infiltration and causes soil saturation (Engels, 2001). Compaction occurs particularly in areas where animals concentrate, such as water points, gates or pathways. Trails can become conduits for surface runoff and can generate new transient streams (Clark Conservation District, 2004; Belsky, Matzke and Uselman, 1999). Increased runoff from uplands results in higher peak flow and increased water velocity. The resulting intensified erosive force increases the level of suspended sediment and deepens the channel. As the channel bed is lowered water drains from the flood plain into the channel, lowering the water table locally. Furthermore, the biogeochemical cycling and the natural ecosystem functions of sediment, nutrient, and biological contaminants can be greatly impaired by excessive water velocity (Rutherford and Nguyen, 2004; Wilcock *et al.*, 2004; Harvey, Conklin and Koelsch, 2003, Belsky, Matzke and Uselman, 1999; Nagle and Clifton, 2003).

In fragile ecosystems such as riparian areas, these impacts can be dramatic. Livestock avoid hot, dry environments and prefer riparian zones because of the availability of water, shade, thermal cover, and the quality and variety of lush verdant forage. A study conducted in the United States (Oregon) showed that riparian areas represent only 1.9 percent of the grazing surface but produced 21 percent of the available forage and contributed 81 percent of forage consumed by cattle (Mosley *et al.*, 1997; Patten *et al.*, 1995; Belsky *et al.*, 1999; Nagle and Clifton, 2003). Cattle, therefore, tend to overgraze these areas and to mechanically destabilize stream banks lowering water availability locally.

Thus we see a whole chain of changes in the riparian environment (see Figure 4.2): riparian hydrology changes – such as lowering groundwater tables, reducing frequencies of over-bank flow and drying out of the riparian zone – are often followed by changes in vegetation and in microbiological activities (Micheli and Kirchner, 2002). A lower water table results in a higher stream bank. As a consequence, the roots of riparian plants are left suspended in drier soils, and the vegetation changes toward xeric species, which do not have the same capacity to protect stream bank and stream water quality (Florinsky *et al.*, 2004). As gravity causes the banks to collapse, the channel begins to fill with sediments. A newly developed low-flow chan-

Figure 4.2 **Process of stream degradation caused by grazing**



Source: after English, Wilson and Pinkerton (1999).

nel begins to form at a lower elevation. The old floodplain becomes a dry terrace, thus lowering water availability throughout the area (see Figure 4.2) (Melvin, 1995; National Public Lands Grazing Campaign, 2004; Micheli and Kirchner, 2002; Belsky *et al.*, 1999; Bull, 1997; Melvin *et al.*, 2004; English, Wilson and Pinkerton,1999; Waters, 1995).

Looking at the potential impact of grazing livestock on the water cycle, particular attention will have to be paid within regions and countries that have developed extensive livestock production systems such as in southern and Central Europe, Central Asia, sub-Saharan Africa, South America, the United States and Australia.

4.4.2 Land-use conversion

As presented in Chapter 2, the livestock sector is an important agent of land conversion. Large areas of original pasture land have been converted into land producing feedcrops. Similarly the conversion of forest to cropland was massive over the last centuries and is still occurring at a fast pace in South America and Central Africa.

A change of land use often leads to changes in the water balance in watersheds, affecting the streamflow,[9] the frequency and level of peak flows, and the level of groundwater recharge. Factors that play a key role in determining the hydrological changes that occur after land use and/or vegetation change include: climate (mostly rainfall); vegetation management; surface infiltration; evapo-transpiration rates of new vegetation and catchment properties (Brown *et al.*, 2005).

Forests play an important role in managing the natural water cycle. The canopy softens the fall of raindrops, leaf litter improves soil infiltration capacity and enhances groundwater recharge. Furthermore, forests and, especially rainforests, make a net demand on streamflow that helped

[9] The stream flow is composed of storm flow (mainly surface runoff) and baseflow (groundwater discharge into the stream) (Zhang and Shilling, 2005).

moderate storm peak flow events over the year (Quinlan Consulting, 2005; Ward and Robinson, 2000 in Quinlan Consulting, 2005). As a result, when forest biomass is removed, total annual water yield usually increases correspondingly.

As long as surface disturbance remains limited, the bulk of the annual increase remains as baseflow. Often, however – especially when grasslands or forests are converted into croplands – rainfall infiltration opportunities are reduced, the intensity and frequency of storm peak flow events are increased, ground water reserves are not adequately replenished during the rainy season, and there are strong declines in dry season flows (Bruijnzeel, 2004). Substantial changes to catchments' runoff are reported after treatments such as the conversion of forest to pasture or the afforestation of grassed catchment (Siriwardena *et al.*, 2006; Brown *et al.*, 2005).

The effects of vegetation composition change on seasonal water yield are highly dependent on local conditions. Brown *et al.* (2005) summarize the expected seasonal response in water yield depending on the types of climate (see Table 4.21). In tropical catchments two types of response are observed: a uniform proportional change over the year, or greater seasonal change during the dry season. In winter-dominant rainfall areas there is a pronounced reduction of summer flows compared to winter flows. This is mainly owing to the fact that rainfall and evapo-transpiration are out of phase: the highest demand for water by vegetation occurs in summer, when water availability is low (Brown *et al.*, 2005).

The case of the Mississippi River Basin perfectly illustrates how land-use conversion related to livestock production affects the seasonal water availability at basin level. In the Mississippi Basin, endogenous cool season plants come out of dormancy in the spring after the soil thaws, go dormant in the heat of the summer and become active again in the fall if not harvested. In contrast, exogenous warm season crops such as corn and soybeans (mainly used as feed) have a growing season that extends over the middle portion of the year. For the latter the peak water demand is reached during mid-summer. The vegetation change in the Mississippi River Basin led to a discrepancy between peak precipitations that occur in spring and early summer and the

Table 4.21

Seasonal effects of vegetation composition change on water yield, by climate type

Climate	Absolute response	Proportional response
Tropical/summer-dominant rainfall	Larger changes in summer months, when rainfall is greater then monthly average	Two patterns of responses observed: (1) Similar changes in all months (2) Larger changes in winter months, when rainfall is below monthly average
Snow-affected catchments	Largest changes in months of of snow melt	Larger change in summer growing season
Winter-dominant rainfall	Largest changes in winter months, when rainfall is above monthly average	Larger changes in summer months when rainfall is below monthly average
Uniform rainfall	Uniform change across all seasons	With deciduous vegetation there is a larger change during the spring months. Evergreen vegetation shows uniform change across all seasons

Note: Absolute response: total volume change over a year.
Proportional response: change with respect to the seasons.
Source: Brown *et al.* (2005).

seasonal water demand of annual crops which peaks in summer. Such human-generated seasonal inadequacy between water supply and demand by the vegetation has greatly influenced the baseflow over the year in this region (Zhang and Schilling, 2005).

4.5 Summary of the impact of livestock on water

Overall, summing up the impacts of all the different segments of the production chain, the livestock sector has an enormous impact on water use, water quality, hydrology and aquatic ecosystems.

The water used by the sector exceeds 8 percent of the global human water use. The major part of this is water used for feed production, representing 7 percent of the global water use. Although it may be of local importance, for example in Botswana or in India, the water used to process products, for drinking and servicing remains insignificant at the global level (below 0.1 percent of the global water use and less than 12.5 percent of water used by the livestock sector) (see Table 4.22).

Evaluating the role of the livestock sector on water depletion is a far more complex process. The volume of water depleted is only assessable for water evapotranspired by feed crops during feed production. This represents a significant share of 15 percent of the water depleted every year.

The volume of water depleted by pollution is not quantifiable, but the strong contribution of the livestock sector to the pollution process has become clear from country-level analysis. In the United States sediments and nutrients are considered to be the main water-polluting agents. The livestock sector is responsible for an estimated 55 percent of erosion and 32 percent and 33 percent, respectively, of the N and P load into freshwater resources. The livestock sector also makes a strong contribution to water pollution by pesticides (37 percent of the pesticides applied in the United States), antibiotics (50 percent of the volume of antibiotics consumed in the United States), and heavy metals (37 percent of the Zn applied on agricultural lands in England and Wales).

Livestock land use and management appear to be the main mechanism through which livestock contribute to the water depletion process. Feed and forage production, manure application on crops, and land occupation by extensive systems are among the main drivers for unsustainable nutrient, pesticide and sediment loads in water resources worldwide. The pollution process is often diffuse and gradual and the resulting impacts on ecosystems are often not noticeable until they become severe. Further, because it is so diffuse, the pollution process is often extremely hard to control, especially when it is taking place in areas of widespread poverty.

The pollution resulting from industrial livestock production (consisting mainly of high nutrients loads, increased BOD and biological contamination) is more acute and more noticeable than from other livestock production systems, especially when it takes place near urban areas. As it impacts human well-being directly, and is easier to control, mitigating the impact of industrial livestock production usually receives more attention from policy-makers.

National and international transfers of virtual water and environmental costs

Livestock production has diverse and complex regional impacts on water use and depletion. These impacts can be assessed through the concept of "virtual water" defined as the volume of water required to produce a given commodity or service (Allan, 2001). For example, on average 990 litres of water are required to produce one litre of milk (Chapagain and Hoekstra, 2004). "Virtual water" is of course not the same as the actual water content of the commodity: only a very small proportion of the virtual water used is actually embodied in the product (e.g. 1 out of 990 litres in the milk example). Virtual water used in various segments of the produc-

Table 4.22

Estimated contribution of the livestock sector to water use and depletion processes

WATER USE			
Dinking and servicing water		Global	0.6% of water use
		United States	1% of water use
		Botswana	23% of water use
Meat and milk processing, tanning		Global	0.1% of water use
Irrigated feed production (excluding forage)		Global	7% of water use
WATER DEPLETION			
Water evapotranspired by feed crops (excluding grassland and forage)		Global	15% of water evapotranspired in agriculture
Nutrient contamination	N	Thailand (pig waste)	14% of N load
		Viet Nam (pig waste)	38% of N load
		China-Guangdong (pig waste)	72% of N load
		United States	33% of N load
	P	Thailand (pig waste)	61% of P load
		Viet Nam (pig waste)	92% of P load
		China-Guangdong (pig waste)	94% of P load
		United States	32% of P load
Biological contamination		N.A.	
Antibiotics consumption		United States	50% of antibiotics consumed
Pesticide (for corn and soybean as feed) applied		United States	37% of pesticides applied
Erosion from agricultural land		United States	55% of erosion process
Heavy metal applied	Zn	England and Wales	37% of Zn applied
	Cu	England and Wales	40% of Cu applied

tion chain can be attributed to specific regions. Virtual water for feed production, destined for intensive livestock production, may be used in a different region or country than water used directly in animal production.

Differences in virtual water used for different segments of livestock production may be related to differences in actual water availability. This partly helps to explain recent trends in the livestock sector (Naylor *et al.*, 2005; Costales, Gerber and Steinfeld, 2006) where there has been an increased spatial segmentation at various scales of the animal food chain, especially the separation of animal and feed production. The latter is already clearly discernable at the national as well as the subnational level when the map of main global feed production areas (Maps 5, 6, 7 and 8, Annex 1) is compared to the distribution of monogastric animal populations (Maps 16 and 17, Annex 1). At the same time, international trade of the final animal products has increased strongly. Both changes lead to increased transport and strongly enhanced global connectivity.

These changes can be considered in the light of the uneven global distribution of water resources. In developing regions, renewable water resources vary from 18 percent of precipitation and incoming flows in the most arid areas (West Asia/North Africa) where precipitation is a mere 180 mm per year, to about 50 percent in humid East Asia, which has a high precipitation of about 1 250 mm per year. Renewable water resources are most abundant in Latin America. National level estimates conceal very wide variations at sub-national level – where environmental impacts actually occur. China, for instance, faces severe water shortages in the north while the south still has abundant water resources. Even a water-abundant country such as Brazil faces shortages in some areas.

Regional specialization and increased trade can be beneficial to water availability in one place, while in another it may be detrimental.

Spatial transfer of commodities (instead of water) theoretically provides a partial solution to water scarcity by releasing pressure on scarcely available water resources at the receiver end. The importance of such flows was first evaluated for the case of the Middle East, i.e. the most water-challenged region in the world, with little freshwater and negligible soil water (Allen, 2003). The livestock sector clearly alleviates this water shortage, via the high virtual water content of the increasing flows of imports of animal products (Chapagain and Hoekstra, 2004; Molden and de Fraiture, 2004). Another strategy for saving local water by using "virtual water" from elsewhere is to import feed for domestic animal production, as in the case of Egypt which imports increasing quantities of maize for feed (Wichelns, 2003). In the future, these virtual flows may significantly increase the impact of the livestock sector on water resources. This is because a great deal of the rapidly increasing demand for animal products is met by intensive production of monogastrics, which relies heavily on the use of water-costly feed.

However, the global flows of virtual water also have an environmental downside. They may even lead to harmful environmental dumping if the environmental externalities are not internalized by the distant producer: in water-scarce regions such as the Middle East the availability of virtual water from other regions has probably slowed the pace of reforms that could improve local water efficiency.

Environmental impacts are becoming less visible to the widening range of stakeholders who share responsibility for them. At the same time, there is the increased difficulty of identifying stakeholders, which complicates the solving of individual environmental issues. For example, Galloway *et al.* (2006) demonstrate that the growing of feed in other countries makes up more than 90 percent of water used for the production of animal products consumed in Japan (3.3 km^3 on a total of 3.6 km^3). Retracing these flows shows that they mainly originate from not particularly water-abundant feed-cropping regions in countries such as Australia, China, Mexico and the United States. Following a similar approach for nitrogen, the authors show that Japanese meat consumers may also be responsible for water pollution in distant countries.

4.6 Mitigation options

Multiple and effective options for mitigation exist in the livestock sector that would allow reversal of current water depletion trends and a move away from the "business as usual" scenario described by Rosegrant, Cai and Cline (2002) of ever increasing water withdrawals and growing water stress and scarcity.

Mitigation options usually rely on three main principles: reduced water use, reduced depletion process and improved replenishment of the water resources. We will examine these in the rest of this chapter in relation to various technical options. The conducive policy environment to support effective implementation of these options will be further developed in Chapter 6.

4.6.1 Improved water-use efficiency

As demonstrated, water use is strongly dominated by the more intensive livestock sector through the production of feed crops, mainly coarse grains and protein-rich oil crops. The options here are similar to those proposed by more generic water and agriculture literature. Though, given the large and increasing share of feed crops in the global consumption of water with substantial opportunity costs, they deserve to be reiterated.

The two main areas with room for improvement are irrigation efficiency[10] and water productivity.

Improving irrigation efficiency

Based on the analysis of 93 developing countries, FAO (2003a) estimated that, on average, irrigation efficiency was around 38 percent in 1997/99, varying from 25 percent in areas of abundant water resources (Latin America) to 40 percent in the West Asia/North Africa region and 44 percent in South Asia where water scarcity calls for higher efficiencies.

In many basins, much of the water thought to be wasted goes to recharge groundwater, or flows back into the river system, so it can be used via wells, of by people and ecosystems downstream. However, even in these situations, improving irrigation efficiency can provide other environmental benefits. In some cases, it can save water — for example if irrigation drainage is flowing into saline aquifers where it can not be reused. It can help prevent agrochemicals from polluting rivers and groundwater; and it can reduce waterlogging and salinization. Many of the actions associated with improving irrigation efficiency can have other advantages. For example:

- canal lining gives irrigation managers more control over water supply;
- water pricing provides cost recovery and accountability; and
- precision irrigation can increase yields and improve water productivity (Molden and de Fraiture, 2004).

In many basins, especially those that are already experiencing water stress, there is little or no irrigation water being wasted, because water recycling and re-use are already widespread. The Nile in Egypt (Molden *et al.*, 1998; Keller *et al.*, 1996), the Gediz in Turkey (GDRS, 2000), the Chao Phraya in Thailand (Molle, 2003), the Bakhra in India (Molden *et al.*, 2001) and the Imperial Valley in California (Keller and Keller 1995), are all documented examples (Molden and de Fraiture, 2004).

Boosting water productivity

Improving water productivity is critical to freeing up water for the natural environment and other users. In its broadest sense, improving water productivity means obtaining more value from each drop of water - whether it is used for agriculture, industry or the environment. Improving irrigated or rainfed agricultural water productivity generally refers to increasing crop yield or economic value per unit of water delivered or depleted. But it can also be extended to include non-crop foods such as fish or livestock. There is a substantial water productivity gain to be obtained from better integration of crop and livestock in mixed systems, particularly by feeding crop residues to livestock, which provide organic fertilizer in return. The potential of this was substantiated for West Africa by Jagtap and Amissah-Arthur (1999). The principle could also be applied to industrialized production systems. While producing corn for often distant monogastrics production sites, large-scale maize-dominated feedcrop areas could easily supply maize residue to local ruminant farms.

Although farms producing feed for industrialized livestock systems generally already operate at relatively high water productivity levels, there may be scope for improvement by for example: selecting appropriate crops and cultivars; better planting methods (e.g. on raised beds); minimum tillage; timely irrigation to synchronize water application with the most sensitive growing periods; nutrient management; drip irrigation and improved drainage for water table control. In

[10] Irrigation efficiency is defined as the ratio between the estimated consumptive water use in irrigation and irrigation water withdrawal (FAO, 2003a).

dry areas, deficit irrigation – applying a limited amount of water but at a critical time – can boost productivity of scarce irrigation water by 10 to 20 percent (Oweis and Hachum 2003).

4.6.2 Better waste management

One of the primary water-related issues that industrialized livestock production systems must face is waste management and disposal. A series of effective technical options, mainly elaborated within developed countries, are already available but they need to be more widely applied and adapted to local conditions within developing countries.

Waste management can be divided into five stages: production, collection, storage, process and utilization. Each stage should be specifically addressed by adequate technological options in order to reduce the livestock sector's current impact on water.

Production stage: a better balanced feed

The production stage refers to the amount and characteristics of faeces and urine generated at the farm level. These vary considerably depending on the composition of the diet, feed management practices, species characteristics and animal growth stages.

Feeding management has improved continuously over the last decades and has resulted in improved production levels. The challenge for producers and nutritionists is to formulate rations that continue to improve production levels while simultaneously minimizing environmental impacts associated with excreta. This can be achieved by optimizing nutrient availability and by better adjusting and synchronizing nutrients and mineral inputs to the animals requirements (e.g. balanced rations and phased feeding), which reduce the quantity of manure excreted per unit of feed and per unit of product. Better feed conversion ratio can also be achieved through animal genetic improvement (Sutton *et al.*, 2001; FAO, 1999c; LPES, 2005).

Dietary strategies to improve feed efficiency rely on four main principles:

- meeting nutrient requirements without exceeding them;
- selecting feed ingredients with readily absorbable nutrients;
- supplementing diets with additives/enzymes/vitamins that improve P availability and guarantee an optimal amino acid supply at reduced crude protein level and retention; and
- reducing stress (LPES, 2005).

Adjusting the diets to the effective requirements has a significant impact on faecal nutrient excretion locally, especially when large animal production units are involved. For example, the level of P in the cattle diet in industrialized systems, generally, exceeds the required level by 25 to 40 percent. The common practice of supplementing cattle diets with P is therefore, in most cases, unnecessary. An adapted diet with adequate P content is, therefore, the simplest way to lower the amount of P excreted by cattle production and has been shown to reduce P excretion in beef production by 40 to 50 percent. Nevertheless, in practice, producers feed cattle with low cost by-products that usually contain high levels of P. Identically, in the United States the usual P content in poultry feed of 450 mg can be lowered to 250 mg per hen per day (National Research Council recommendation) without any production loss and with valuable feed savings (LPES, 2005; Sutton *et al.*, 2001).

Similarly the content of heavy metals in manure can be reduced if an appropriate diet is provided. Successful examples have proved the efficiency of this measure. In Switzerland the mean (median) content of Cu and Zn in pig manures decreased considerably between 1990 and 1995 (by 28 percent for Cu, 17 percent for Zn) demonstrating the effectiveness of limiting heavy metals in animal feed to required levels (Menzi and Kessler, 1998).

Modifying the balance of feed components and the origin of the nutrients can significantly influence nutrient excretion levels. For cattle, a proper balance in feed between degradable and non-degradable proteins improves nutrient absorption and has been shown to reduce N excretion by 15 to 30 percent without affecting production levels. Nevertheless, this is usually linked to an increase in the proportion of concentrate in the ration, which on grassland farms means a decreased use of own roughage resulting in extra costs and nutrient balance surplus. Similarly, adequate levels of carbohydrate complexes, oligosaccharides and other non-starch polysaccharides (NSP) in the diet can influence the form of N excreted. They generally favour the production of bacterial protein that is less harmful to the environment and has a higher recycling potential. For pigs a lower amount of crude protein supplemented with synthetic amino acids lowers N excretion up to 30 percent, depending on the initial composition of the diet. Similarly, in pig production systems, the quality of feed plays an important role. Removing fibre and germ from corn is reported to reduce the level of dry matter excreted by 56 percent and the level of N contained in urine and the faeces by 39 percent. Using organic forms of Cu, Fe, Mn and Zn in swine diets reduces the level of heavy metals added to the ration and significantly reduces excretion levels without depressing growth or feed efficiency (LPES, 2005; Sutton *et al.*, 2001).

In order to improve feed efficiency new sources of highly digestible feedstuff are being developed through classical breeding techniques or genetic modification. The two main examples reported are the development of low-phytate corn, which reduces P excretion, and of low stachyose soybeans. P availability in classical feed (corn and soybean) is low for pigs and poultry as P is usually bound in a phytate molecule (90 percent of the P in corn is present in phytate form, and 75 percent in soybean meal). This low P availability is because phytase, which can degrade the phytate molecule and make P available, is lacking in the digestive systems of pigs and poultry. The use of low phytate P genotypes reduces the levels of mineral P to be supplemented in the diet and reduces P excretion by 25 to 35 percent (FAO, 1999c; LPES, 2005; Sutton *et al.*, 2001).

Phytase, xylanase and betaglucanase (which are also not naturally excreted by pigs) could be added to feed in order to favour degradation of non-starch polysaccharides available in cereals. These non-starch polysaccharides are usually associated with protein and minerals. The absence of such enzymes results in lower feed efficiency and increases mineral excretion. The use of phytase has been shown to improve P digestibility in pig diet by 30 to 50 percent. Boling *et al.* (2000) achieved a 50 percent reduction in faecal P content from laying hens by providing a low P diet supplemented with phytase, together with the maintenance of an optimal egg production level. Similarly, addition of 1.25 dihydroxy vitamin D3 to broiler feed reduced phytate P excretion by 35 percent (LPES, 2005; Sutton *et al.*, 2001).

Other technological improvements include particle reduction, pelleting and expanding. Particle size of 700 microns is recommended for better digestibility. Pelleting improves feed efficiency by 8.5 percent.

Finally, improving animal genetics and minimizing animal stress (adapted brooding, ventilation and animal health measures) improves weight gain and, therefore, feed efficiency (FAO, 1999c; LPES, 2005).

Improving manure collection process

The collection stage refers to the initial capture and gathering of the manure at the point of origin (see Figure 4.3). The type of manure produced and its characteristics are greatly affected by collection methods used and the amount of water added to the manure.

Animal housing has to be designed to reduce losses of manure and nutrients through runoff. The type of surface on which animals are grown is one of the key elements that influence the collection process. A slatted floor can greatly facilitate immediate manure collection, but it implies that all the excreta are collected in liquid form.

Contaminated runoff from production areas should be redirected into manure storage facilities for processing. The amount of water used in the animal house and originating from rainfall (especially in warm and humid areas) entering in contact with manure should be reduced to its minimum to limit the dilution process which, otherwise, increases the volume of waste (LPES, 2005).

Improved manure storage

The storage stage refers to the temporary containment of manure. The storage facility of a manure management system gives the manager control over the scheduling and timing of the system functions. For example it allows timely application on the field in accordance with the nutrient requirements of the crops.

Improved manure storage aims to reduce and ultimately prevent leakage of nutrients and minerals from animal housing and manure storage into groundwater and surface water (FAO, 1999c). Appropriate storage capacity is of prime importance to prevent losses through overflow, especially during the rainy season in tropical climates.

Improved manure processing

Technical options for manure processing exist that can reduce the potential for pollution, reduce local manure surpluses and convert surplus manure in products of higher value and/or products that are easier to transport (including biogas, fertilizer and feed for cattle and fish). Most of the technologies aim at concentrating the nutrients derived from separated solids, biomass or sludge (LPES, 2005; FAO, 1999c).

Manure processing includes different technologies that can be combined. These technologies include physical, biological and chemical treatment and are presented in Figure 4.3.

Transport of unprocessed litter, or manure, over long distances is impractical because of the weight, cost and the unstable properties of the product. The initial step in manure processing usually consists of separation of solids and liquids. Basins can be used to allow the sedimentation process and facilitate removals of solids from feedlot runoff, or before lagooning. Smaller solids can be removed in a tank where water velocity is greatly reduced. However, sedimentation tanks are not used often for animal manure, as they are costly. Other technologies for removal of solids include incline screens, self-cleaning screens, presses, centrifuge-type processes and rapid sand filters. These processes can reduce significantly the loads of C, N and P in subsequent water flows (LPES, 2005).

The choice of the initial step is of prime importance as it greatly influences the value of the final product. Solid wastes have low handling costs, lower environmental impact potential and a higher market value as nutrients are concentrated. In contrast, liquid wastes have lower market value as they have high handling and storage costs, and their nutrient value is poor and unreliable (LPES, 2005). Furthermore, liquid waste has a much higher potential to impact the environment if storing structures are not impermeable or do not have a sufficient storage capacity.

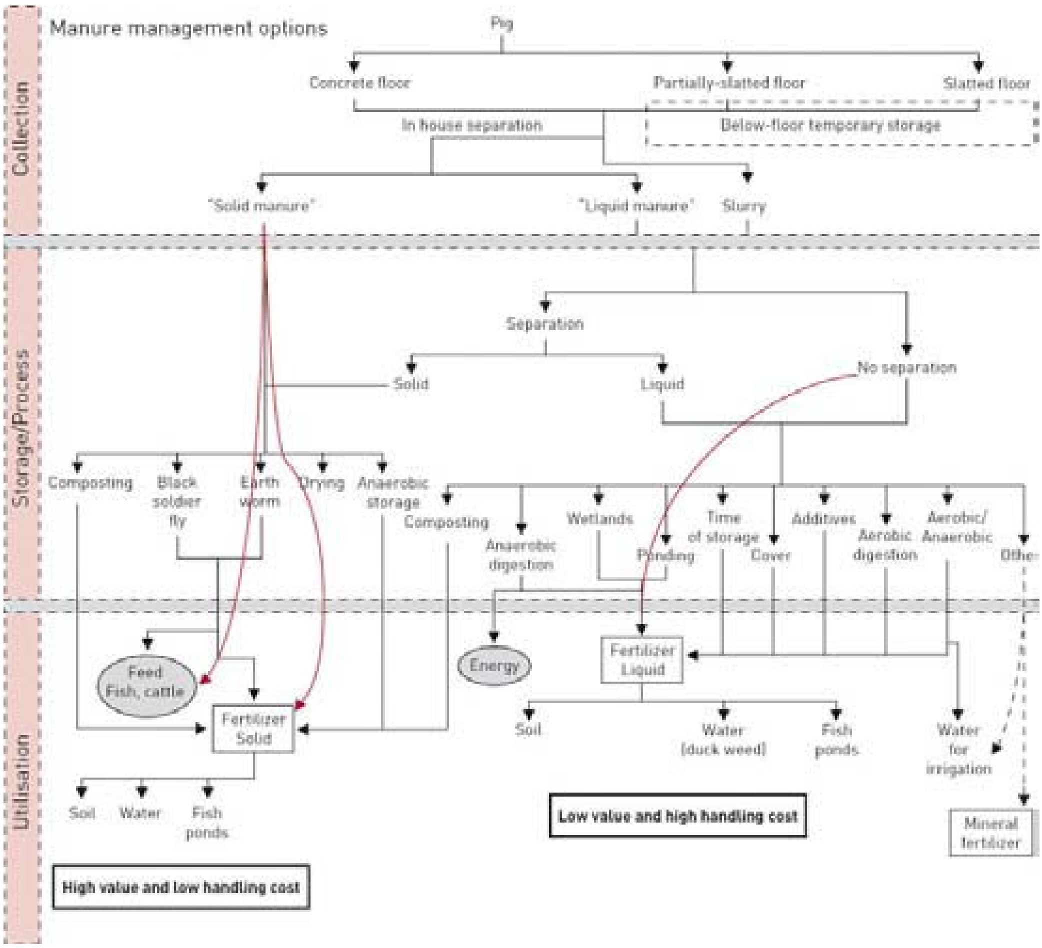
As presented in Figure 4.3 the separation phase can be followed by a wide range of optional processes that influence the nature of the final product.

Classical technical options already in widespread use include:

Aeration: This treatment removes organic material and reduces the biological and chemical oxygen demand. 50 percent of the C is converted into sludge or biomass which is collected by sedimentation. P is also reduced by biological uptake but to a lesser extent. Different types of aerobic treatment can be used, such as activated sludge[11] (where the biomass returns to the inflow

[11] The activated sludge process uses the organic matter of wastewater to produce a mixed population of microorganisms, in an aerobic environment.

Figure 4.3 **Technical options in manure management**



Source: FAO (2003b).

portion of the basin) or trickling filters in which the biomass grows on a rock filter. Depending on the depth of the lagoon, aeration can be applied to the entire volume of lagooning systems or to a limited to a portion of it to benefit from aerobic and anaerobic digestion processes simultaneously (LPES, 2005).

Anaerobic digestion: The major benefits of an anaerobic digestion process and the reduction of chemical oxygen demand (COD), biological oxygen demand (BOD) and solids, and the production of methane gas. Nevertheless it does not reduce N and P contents (LPES, 2005).

Sedimentation of biosolids: The generated biomass is treated biologically in sedimentation tanks or clarifiers, in which water flow velocity is slow enough to allow solids above a certain size or weight to be deposited (LPES, 2005).

Flocculation: The addition of chemicals can improve the removal of solids and dissolved elements. The most common chemicals include lime, alum and polymers. When lime is used, the resulting sludge can have enhanced agronomic value (LPES, 2005)

Composting: Composting is a natural aerobic process that allows the return of nutrients to the soil for future use. Composting usually requires the addition of a substrate rich in fibre and carbon to animal excreta. In some systems innocula and enzymes are added to aid the composting process. Engineered systems that convert manure into a value-added marketable product have become increasingly popular. The benefits of composting are numerous: available organic matter is stabilized and no longer decomposable, odours are reduced to acceptable levels for land application, volume is reduced by 25 to 50 percent and germs and seeds are destroyed by the heat generated by the aerobic formation phase (around 60°C). If the original C:N ratio is above 30, most of the N is conserved during this process (LPES, 2005).

Drying of solid manure is also an option to reduce the volume of manure to be transported and to increase the nutrient concentration. In hot climates, natural drying is possible with at minimal costs outside of the rainy period.

Different processes can be integrated within a single structure. In ***lagooning systems*** the manure is highly diluted, which favours natural biological activity and hence reduces pollution. Effluents can be removed through irrigation to crops which recycle the excess nutrients. Anaerobic lagoon designs work better in warm climates, where bacteriological activity is maintained throughout the year. Anaerobic digesters, with controlled temperature, can be used to produce biogas and reduce pathogens, though they require high capital investments and high management capacity. Nevertheless, most lagooning systems have poor efficiency regarding P and N recovery. Up to 80 percent of all N entering into the system is not recovered but most of the atmospheric release of nitrogen may be in the form of harmless N_2 gas. Most of the P will be recovered only after 10 to 20 years, when the sludge has to be removed. As a result N and P recovery are not synchronized. Lagoon effluent should, therefore, be used primarily as nitrogen fertilizer. The management of the effluent also requires expensive irrigation equipment for what is actually a low-quality fertilizer. The size of the lagoon should be proportional to farm size, which also limits the adoption of the technology as it require large areas for implementation (Hamilton *et al.*, 2001; Lorimor *et al.*, 2001).

Alternative technologies need further research and development to improve their efficiencies and effectiveness: they include chemical amendments, wetland treatment or digestion by worms (Lorimor *et al.*, 2001). Wetland systems are based on the natural nutrient recycling capacities occurring in wetland ecosystems or riparian areas, and have a high potential for removing high levels of N. Vermicomposting is a process by which manure is transformed by earthworms and micro-organisms into a nutrient-rich humus called vermicompost in which nutrients are stabilized (LPES, 2005).

In order to be economically and technologically viable, most processes require large quantities of manure and are generally not technically suitable for implementation on most farms. The feasibility of large- and medium-scale manure processing also depends on local conditions (local legislation, fertilizer prices) and processing costs. Some of the end-products have to be produced in very large quantities and must be of a very reliable quality before being accepted by the industry (FAO, 1999c).

Improved utilization of manure

Utilization refers to the recycling of reusable waste products, or the reintroduction of non-reusable waste products into the environment.

Most often manure is used in the form of fertilizer for agricultural lands. Other uses include feed production (for fish in aquaculture), energy (methane gas), or algal growth fertilizer. Ultimately the nutrients lost could be recycled and reused as feed additives. For example, it has been shown experimentally that layer manure settled in lagoons can serve after processing as a source of calcium and phosphorus, and be re-fed to hens or poultry without impacting production levels (LPES, 2005).

From an environmental point of view, application of manure to cropland or pastures reduces the requirements of mineral fertilizer. Manure also increases soil organic matter, improves soil structure, fertility and stability, reduces soil vulnerability to erosion, improves water infiltration and the water-holding capacity of the soil (LPES, 2005; FAO, 1999c).

Nevertheless, some aspects have to be carefully monitored during the application of organic fertilizers, in particular the level of runoff, which might contaminate freshwater resources, or the build up of excessive nutrient levels in soils. Furthermore, organic N can also be mineralized at times with low N uptake of crops and then be prone to leaching. Environmental risks are reduced if lands are manured with the right method, at adequate application rates, during the right period, and at the right frequency and if spatial characteristics are taken into account.

Practices that limit soil erosion and runoff or leaching or which limit the build-up of nutrients levels in soil include:

- Dosing of fertilizers and manure in agreement with crop requirements.
- Avoiding soil compaction and other damages through soil tillage which might impede the water absorbing capacity of the soil.
- Phytoremediation: selected plant species bioaccumulate the nutrients and heavy metal added to the soil. Bioaccumulation is improved when crops have deep roots to recover subsurface nitrates. The growing of high biomass plants can remove large amounts of nutrients and reduce nutrients levels in soils. The bioconcentration capacity for nutrients and heavy metals varies depending on plant species and varieties.
- Soil amendment with chemicals or municipal by-products, to immobilize P and heavy metals. Soil amendment has already proved to be very effective, and can reduce the discharge of P via runoff water by 70 percent. Soil amendment with polymeric sediment flocculants (such as polyacrylamide polymers) is a promising technology for reducing the transport of sediment and particulate nutrients.
- Deep tillage to dilute nutrient concentration in the near-surface zone.
- Development of strip cropping, terraces, vegetated water ways, narrow grass hedges and vegetative buffer strips, to limit run off and increase the filtration levels of nutrients, sediments and heavy metals (Risse *et al.*, 2001; Zhang *et al.*, 2001).

Despite the advantage of organic fertilizers (e.g. maintenance of soil organic matters), farmers often prefer mineral fertilizers, which guarantee nutrient availability and are easier to handle. In organic fertilizers, nutrient availability varies with climate, farming practices, animal diets and waste management practices. Furthermore where animal production is geographically concentrated, the affordably accessible land for manure application at an adequate rate is usually insufficient. The cost related to manure storage, transport, handling and processing limits the economic viability of using this recycling process further afield by exporting manure from surplus to deficit areas. The processing and transport of manure is viable from an economic viewpoint on the larger scale. Technologies such as separation, screening, dewatering and condensing that reduce the costs associated with the recycling process (mainly storage and transport) should be improved and the right incentives should be developed to favour their adoption (Risse *et al.*, 2001).

4.6.3 Land management

The impacts of extensive livestock production systems on watersheds depend strongly on how grazing activities are managed. Farmers' decisions influence many parameters that affect vegetation change, such as grazing pressure (stocking rate and intensity) and the grazing system (which influences the distribution of animals). The proper control of grazing season, intensity, frequency and distribution can improve vegetation cover, reduce erosion and as a result, can maintain or improve water quality and availability (FAO, 1999c; Harper *et al.*, 1996; Mosley *et al.*, 1997).

Adapted grazing systems, range improvement and identification of critical grazing period

Rotational grazing systems can mitigate impacts to riparian areas by reducing the length of time the area is occupied by cattle (Mosley *et al.*, 1997). Research results on the effect of rotational cattle grazing efficiency on riparian conditions are controversial. Nevertheless, streambank stability has been shown to improve when a rest rotation grazing system replaced heavy, season-long grazing (Mosley *et al.*, 1997; Myers and Swanson, 1995).

The resilience of different ecosystems to cattle impacts differs, depending on soil moisture, plant species composition and animal behaviour patterns. The identification of the critical period is of prime importance in order to design adapted grazing plans (Mosley *et al.*, 1997). For example, stream banks are more easily broken during the rainy season, when soils are wet and susceptible to trampling and sloughing or when excessive browsing may damage vegetation. These impacts can often be reduced if the natural foraging behaviour of cattle is considered. Cattle avoid grazing excessively cold or wet sites and may prefer upland forage when it is more palatable than forage in riparian areas (Mosley *et al.*, 1997).

Trails can be constructed to ease the access to farms, ranches and field. Livestock trails also improve livestock distribution (Harper, George and Tate, 1996). Improved access reduces soil trampling and the formation of gullies that accelerate erosion. With a little training, well-designed hardened crossings often turn into a preferred access point for livestock. This can reduce impact along most of a stream by reducing bank sloughing and sediment inputs (Salmon Nation, 2004). Grade stabilization practices can be used to stabilize the soil, control the erosion process and limit the formation of artificial channels and gullies. Well-located basins can collect and store debris and sediments from water which is passed downstream (Harper, George and Tate, 1996).

Improving livestock distribution: exclusion and other methods

Exclusion of livestock is the key method for recovery and protection of an ecosystem. Animals congregating near surface water increase water depletion, mainly through direct discharge of waste and sediment into water, but also indirectly by reducing infiltration and increasing erosion. Any practise that reduces the amount of time cattle spend in a stream or near other water points, and hence reduces trampling and manure loading, decreases the potential for adverse effects of water pollution from grazing livestock (Larsen, 1996). This strategy can be associated with livestock parasite control programmes to reduce the potential for biological contamination.

Several management practices have been designed in order to control or influence livestock distribution and to prevent cattle from congregating near surface waters. These methods include exclusionary methods such as fencing and the development of buffer strips near surface water, as well as more passive methods that influence cattle distribution such as:

- development of off-stream watering;
- strategically distributed points for supplemental feeds and minerals;
- fertilizer and reseeding activities;
- predator and parasite controls that may hinder the use of some part of the land;
- prescribed burning; and
- trail building.

However, few of these have been widely tested in the field (Mosley *et al.*, 1997).

The time spent by livestock in or very near water has a direct influence on both the deposition and re-suspension of microbes, nutrients and sediment and thus on the occurrence and extent of downstream pollution of water. When livestock are excluded from areas surrounding water resources, direct deposition of livestock waste into water is limited (California trout, 2004; Tripp *et al.*, 2001).

Fencing is the simplest way to exclude livestock from sensitive areas. Fencing activities allow farmers to designate separate pastures that can be managed for recovery, or where limited grazing can occur. Extended periods of rest or deferment from grazing may be needed to enable badly degraded sites to recover (California trout, 2004; Mosley *et al.*, 1997). Fences can be used in order to prevent direct deposition of faeces into water. Fences should be adapted, in terms of size and materials, so as not to impede wildlife activity. For example, the top wire on both riparian pastures and riparian enclosures should not be barbed because riparian areas provide big-game habitat and water for surrounding uplands (Salmon Nation, 2004; Chamberlain and Doverspike, 2001; Harper, George and Tate, 1996).

Recent efforts to improve the health of riparian areas have focussed on the establishment of conservation buffers, to exclude livestock from areas surrounding surface water resources (Chapman and Ribic, 2002). Conservation buffers are strips of land along freshwater courses under permanent, relatively undisturbed vegetation. They are designed to slow water runoff, remove pollutants (sediments, nutrients, biological contaminants and pesticides), improve infiltration and to stabilize riparian areas (Barrios, 2002; National Conservation Buffer Team, 2003; Tripp *et al.*, 2001; Mosley *et al.*, 1997).

When strategically distributed over the agricultural landscape (which may include some parts of the catchment areas), buffers can filter and remove pollutants before they reach streams and lakes or leach to deep ground water resources. The filtering process is mainly the result of an increased frictional process and decreased water velocity of surface runoff. Buffers enhance infiltration, deposition of suspended solids, adsorption to plant and soil surfaces, absorption of soluble material by plants, and microbial activity. Buffers also stabilize stream banks and soil surfaces, reduce wind and water velocity, reduce erosion, reduce downstream flooding and increase vegetation cover. This leads to improved stream habitats for both fish and invertebrates (Barrios, 2002; National Conservation Buffer Team, 2003; Tripp *et al.*, 2001; Mosley *et al.*, 1997; Vought *et al.*, 1995).

Conservation buffers are generally less expensive to install than practices requiring extensive engineering and costly construction methods (National Conservation Buffer Team, 2003). Nevertheless, farmers have often considered them impractical (Chapman and Ribic, 2002) as they restrain access to luxuriant areas that farmers consider crucial for animal production and health especially in dryland areas.

When there is a large stream-to-land-area ratio, preventing faecal deposition into streams by fencing out livestock can become very costly. Providing alternative drinking sources may reduce the time animals spend in the stream and, therefore, the in-stream faecal deposition. This cost effective technical option also improves cattle distribution and reduces the pressure on riparian areas. An off-stream water source has been shown to reduce the amount of time a group of hay-fed animals spent in the stream by more that 90 percent (Miner *et al.*, 1996). Furthermore, even when the source of feed was placed at equal distance between the water tank and the stream, the water tank was still effective in reducing the amount of time cattle spent in the stream (Tripp *et al.*, 2001; Godwin and Miner, 1996; Larsen, 1996; Miner *et al.*, 1996).

The development of water dams, boreholes and watering points should be carefully planned to limit the impact of local concentrations of animals. To avoid degradation by animals, measures for protection of water storage are useful. The reduction of water loss by infiltration can be done by using impermeable materials. Other measures (such as anti-evaporation covers: plastic film, neutral oil) should be implemented to reduce loss through evaporation which is very substantial in hot countries. Nevertheless, the technical options available to limit evaporation are usually expensive and difficult to maintain (FAO, 1999c).

Fertilization can be used as a method of controlling livestock grazing distribution. On foothill rangeland in central California (United States), fertilizing adjacent slopes with sulphur (S) led to significant decreases in the amount of time cattle spent grazing in moist depressions during the dry season (Green *et al.*, 1958 in Mosley *et al.*, 1997).

Providing supplemental feed may also attract livestock away from surface waters. Ares (1953) found that cottonseed meal mixed with salt successfully distributed cattle away from water sources on desert grassland in south central New Mexico. Nevertheless it seems that salt placement is generally incapable of overriding the attraction of water, shade and palatable forage found in riparian zones (Vallentine, 1990). Bryant (1982) and Gillen *et al.* (1984) reported that salting alone was largely ineffective in reducing cattle use of riparian zones. (Mosley *et al.*, 1997)

During dry and hot season livestock tend to spend more time in riparian areas. One technical option is to provide alternative sources of shade away from fragile areas and freshwater resources (Salmon Nation, 2004).

As presented in this section, a large number of technical options are available to minimize the impacts of the livestock sector on water resources, limiting water depletion trends and improving water use efficiency. Nevertheless, these technical options are not widely applied because: a) practices having an impact on water resources are usually more "costs effective" in the short term; b) there is a clear lack of technical knowledge and information dissemination; c) there is a lack of environmental standards and policies and/or their implementation is deficient. In most cases the adoption of adapted technical options reducing water depletion trends will only be achieved through the design and implementation of an appropriate policy framework as presented in Chapter 6.

05

Livestock's impact on biodiversity

5.1 Issues and trends

An unprecedented crisis

Biodiversity refers to the variety of genes, species and ecosystems that can be found in the environment. Short for biological diversity, the term encompasses the entire expression for life on the planet and is generally categorized in three dimensions:

- genetic diversity or the total of genetic information contained in the genes of individual plants, animals and micro organisms;
- species diversity or the variety of living organisms on earth; and
- ecosystem diversity or the variety of habitats and ecological processes in the biosphere.

Biodiversity contributes to many constituents of human well-being, including security, basic materials for a good life, health, good social relations and freedom of choice and action (MEA, 2005b). It does so directly (through provisioning, regulating and cultural ecosystem services) and indirectly (through supporting ecosystem services). Biodiverse ecosystems tend to be more resilient and can therefore better cope with an increasingly unpredictable world (CBD, 2006). For centuries, human beings have benefited from the exploitation of biodiversity, at the same time as they were often reducing it by conversion of natural ecosystems for human uses. Agriculture, livestock, fisheries and forestry have placed significant pressures on biodiversity while providing the basic building blocks for development and economic growth.

The world's biodiversity is facing a crisis without precedence since the end of the last ice age, affecting all its three dimensions. Genetic diversity is at risk, as wild population sizes shrink drastically and with them the gene pool. Species diversity is confronted with rates of extinction that far exceed the "background rate" found in the typical fossil record. The full range of ecosystems diversity is being threatened by transformation through human activities.

The millennium ecosystem assessment (MEA) examined the state of 24 ecosystem services that make a direct contribution to human well-being. It concluded that 15 out of 24 are in decline. And as the Global Biodiversity Outlook of the Convention on Biological Diversity points out, there are important additional reasons to care about the loss of biodiversity, quite apart from nature's immediate usefulness to humankind. Future generations have a right to inherit a planet thriving with life, and which continues to afford opportunities to reap the economic, cultural and spiritual benefits of nature (CBD, 2006). Many would argue that every life form has an intrinsic right to exist. Species alive today are millions of years old and have each traveled unique evolutionary paths, never to be repeated, in order to reach their present form.

Concern over the loss of biodiversity, and the recognition of its crucial role in supporting human life, led to the creation, in 1992, of the Convention on Biological Diversity (CBD) a legally binding global treaty having the objective of the conservation of biodiversity and the sustainable use of its components. As important tools, the CBD includes the development of national biodiversity strategies and action plans. Although almost every country developed such strategies, progress remains very limited towards essential goals such as the improvement of capacity for implementation and national-level planning, as well as actual implementation (CBD, 2006). The greatest conservation efforts are pursued over endangered species and their habitats, while ecosystems services receive less consideration.

According to the MEA Report (2005b), the most important direct drivers of biodiversity loss and ecosystem service changes are:

- habitat change (such as land use changes, physical modification of rivers or water withdrawal from rivers, loss of coral reefs, and damage to sea floors resulting from trawling);
- climate change;
- invasive alien species;
- overexploitation; and
- pollution.

Livestock play an important role in the current biodiversity crisis, as they contribute directly or indirectly to all these drivers of biodiversity loss, at the local and global level. Typically, biodiversity loss is caused by a combination of various processes of environmental degradation. This makes it hard to single out the contribution of the livestock sector, and this is further complicated by the many steps in the animal food product chain at which environmental impact occurs.

Livestock-related land use and land-use change modify or destroy ecosystems that are the habitats for given species (see Chapter 2). Livestock contribute to climate change, which in turn has a modifying impact on ecosystems and species (see Chapter 3). Terrestrial and aquatic ecosystems are affected by emissions into the environment (nutrient and pathogen discharge in marine and freshwater ecosystems, ammonia emissions, acid rain). The sector also directly affects biodiversity through invasive alien species (the livestock themselves and diseases for which they may be vectors) and overexploitation, for example through overgrazing of pasture plants. This complex picture is further complicated by the fact that livestock first started to affect biodiversity millennia ago when they were domesticated and provided humans with a way to exploit new resources and territories that were previously unavailable. These historic changes continue to affect biodiversity, while the effect of current degradation processes (many of which are described in the preceding chapters) is superimposed.

This chapter first provides an overview of the status of global biodiversity. Then livestock's contribution to biodiversity loss is assessed, along the various steps of the animal product food chain. As a consequence of the complexity described above, this assessment is sometimes necessarily fragmentary and anecdotal, but it still provides an indication not only of the significance of the livestock sector's impact but also of the challenges of – and opportunities for – slowing, halting or reversing the degradation process. A number of technical options exist to reduce the negative impact of a number of some current practices and change processes. These options are presented in a last section.

5.2 Dimensions of biodiversity

Biodiversity is characterized by multiple dimensions. At the level of living organisms intra- and inter-species diversity mostly refers to the genetic and phenotypic side of biodiversity. At higher scales biodiversity through ecosystem richness refers to how species are assembled into diverse biotic communities within a wide range of biotopes.[1]

Inter-species diversity

Inter-species biodiversity refers to the total number of species (animals, plants and microbes) on earth. The total number of species is still unknown. Around 1.8 million species have been described to date, but many more are believed to exist – estimates range from 5 to nearly 100 million. 14 million has been proposed as a reasonable working estimate (see Table 5.1). Based on the latter figure, only 12 percent of the estimated total number of species have been classified so far.

Table 5.1

Estimated numbers of described species, and possible global total

Kingdoms	Described species	Estimated total species
Bacteria	4 000	1 000 000
Protoctists (algae, protozoa, etc.)	80 000	600 000
Animals	1 320 000	10 600 000
Fungi	70 000	1 500 000
Plants	270 000	300 000
Total	1 744 000	14 000 000

Source: UNEP-WCMC (2000).

Existing species are not evenly distributed on the globe. Some areas are much richer in species than others and many species are endemic to a specific region. In general diversity declines towards the poles. Humid tropical regions are particularly rich in species and contain numerous endemic ones. The environments richest in biodiversity are moist tropical forests that extend over some 8 percent of the world's land surface, yet they hold more than 50 percent of the world's species. Tropical regions support two-thirds of the estimated 250 000 plant species, and 30 percent of the bird species. Similarly, inland waters represent a vanishing small proportion of the earth's total water but they contain 40 percent of all aquatic species that are often endemic (Harvey, 2001).

Intra-species diversity

Intra-species diversity refers to richness of genes within a given species. It encompasses the genetic variation among individuals within the same population and among populations. Genetic diversity represents a mechanism for populations and species to adapt to changing

[1] A biotope is an area that is uniform in environmental conditions and in its distribution of animal and plant life.

environments. Intra-species diversity is crucial to the resilience of populations and ecosystems against unpredictable and random events. The greater the variation, the higher the chances that a species will have some individuals with genes adapted for a new environment that can be passed on to the next generation. Reduced intra-species diversity not only reduces resilience, but also increases the probability of inbreeding, often leading to an increase in genetic diseases that may in the long run threaten the species itself.

The best known example of intra-species diversity is in agricultural biodiversity. Agricultural biodiversity is a creation of humankind and includes domesticated plants and animals, as well as non-harvested species that support food provision within agro-ecosystems. In the case of livestock, the initial natural selection that gave birth to the wild progenitor was followed by thousands of years of domestication and selective breeding by humans. Farmers and breeders have selected animals for a variety of traits and production environments, resulting in the development of more than 7 600 breeds of livestock (FAO, 2006c). From just nine of the 14 most important species (cattle, horse, ass, pig, sheep, buffalo, goat, chicken and duck) as many as 4 000 breeds have been developed and used worldwide.

In the wild, intra-species genetic diversity is becoming a central preoccupation for wildlife management and conservation. When populations become too isolated, inbreeding phenomena may result if the size of the population is not large enough. Therefore, allowing isolated wildlife populations to interbreed can help exchange of genes and improve the genetic pool of wildlife populations.

Ecosystem diversity

An ecosystem is an assemblage of living species within a biotope that through the interaction with its physical environment functions as a unit. Most classification systems for ecosystems use biological, geological and climate characteristics, including topography, vegetation cover or structure, even cultural or anthropogenic factors. Ecosystems can be of any scale, from a small pond to the entire biosphere, and interact with each other.

Attempts have been made to characterize ecosystems and their diversity over wide areas. WWF (2005) defines an ecoregion as a large area of land or water that contains a geographically distinct assemblage of natural communities that (a) share a large majority of their species and ecological dynamics; (b) share similar environmental conditions, and; (c) interact ecologically in ways that are critical for their long-term persistence. Using this approach, WWF has identified 825 terrestrial ecoregions globally (a set of approximately 500 freshwater ecoregions is under development) and assessed the status of ecosystem diversity in each of these regions. On a still broader scale the World Resources Institute (2000) distinguishes five major and critical biomes shaped by the interaction of physical environment, biological conditions and human intervention: agro-ecosystems, coastal and marine ecosystems, forest ecosystems, freshwater systems and, grassland ecosystems. Forests, which harbour about two-thirds of the known terrestrial species, have the highest species diversity and local endemism of any biome.

Ecosystems are central to the functioning of the planet as they provide services that regulate the main natural cycles (water, carbon, nitrogen, etc.). These services include: maintenance of watershed functions (infiltration, flow and storm control, soil protection); pollution removal from air and water (including the recycling and sequestration of carbon, nutrients and chemical pollutants); and provision of habitat for wildlife. For humans, ecosystems provide a wide range of goods and services including food, energy, materials and water, but also aesthetic, cultural and recreational values. The level of goods and services provided vary greatly between the different ecosystems.

Biodiversity under threat[2]

The three dimensions of biodiversity (genes, species and ecosystems) are all interconnected, and all are eroding at a fast pace worldwide. Any phenomenon impacting one dimension will inevitably impact the others: reduction of genetic diversity can lead, at the extreme, to local or total extinction of a species. The disappearance of one species can break the balance between the different wildlife population species, which may in turn affect ecosystem functioning: predators have been shown to be critical to diversity and stability. For example, the hunting of carnivores has often resulted in increased herbivore populations leading to changes in vegetation affecting many species. Similarly, habitat destruction, change and fragmentation threaten intra- and inter-species genetic diversity. This occurs first because the total area and carrying capacity of the wildlife habitat is reduced by the conversion process, and second because fragmented habitats isolate populations from another, narrowing the genetic pool of each population and making them more vulnerable to disappearance.

The principal threats by ecosystem are presented in Table 5.2. Forested ecosystems, and in particular primary forest ecosystems, are under great threat at the global level. Global forest cover has been reduced by 20 and 50 percent since pre-agricultural times (Matthews *et al.*, 2000). As much as 30 percent of the potential area of temperate, subtropical and tropical forests has been converted to agriculture. Since 1980 forest area has increased slightly in industrial countries, but has declined by almost 10 percent in developing countries (WRI, 2000). The great majority of forests in industrial countries, except Canada and the Russian Federation, are reported to be secondary forest (having regrown after being logged over at least once) or converted to plantations. These areas are poor in biodiversity, compared to the original primary forest, and the loss of many species during the land-use transition is often final. Tropical deforestation affecting primary forest probably exceeds 130 000 km^2 a year (WRI, 2000).

The world's freshwater systems are so degraded that their ability to support human, plant and animal life is greatly imperiled. Half the world's wetlands are estimated to have been lost in the twentieth century, converted to agriculture and urban areas, or filled and drained to combat diseases such as malaria. As a result, many freshwater species are facing rapid population decline or extinction and freshwater resources for human use are increasingly scarce.

The conversion of coastal ecosystems to agriculture and aquaculture, along with other pressures such as erosion and pollution, are reducing mangroves, coastal wetlands, sea grass areas and coral reefs at an alarming rate. Coastal ecosystems have already lost much of their capacity to produce fish because of over-fishing, destructive fishing techniques and destruction of nursery habitats.

Temperate grasslands, savannahs, and shrublands have experienced heavy conversion to agriculture, more so than other grassland types including tropical and subtropical grasslands, savannahs and woodlands. In many places the introduction of non-native species has negatively affected grassland ecosystems leading to a decrease in biodiversity.

Agro-ecosystems are also under great threat. Over the last 50 years, about 85 percent of the world's agricultural land has been affected to some degree by degradation processes including erosion, salinization, compaction, nutrient depletion, biological degradation and pollution. About 34 percent of all agricultural land contains areas only lightly degraded, 43 percent contains moderately degraded areas and nine percent contains strongly or extremely degraded areas (WRI, 2000). Agricultural intensification often diminishes biodiversity in agricultural areas, for example through the excessive application of fertilizer and pesticides, by reducing the

[2] Drawn from UNDP, UNEP, WB and WRI (2000); and Baillie, Hilton-Taylor and Stuart, 2004.

Table 5.2
Major ecosystems and threats

Categories	Major ecosystems	Major threats
Marine and coastal	Mangroves, coral reefs, sea grasses, algae, pelagic communities, deep sea communities	Chemical pollution and eutrophication, overfishing, global climate change, alterations of physical habitat, invasion of exotic species.
Inland water	Rivers, lakes, wetlands (bogs, fens, marshes, swamps)	Physical alteration and destruction of habitat through water extraction, drainage, canalization, flood control systems, dams and reservoirs, sedimentation, introduced species, pollution (eutrophication, acid deposition, salinization, heavy metals).
Forest	Boreal and temperate cornifers, temperate broadleaf and mixed, tropical moist, tropical dry, sparse trees and parkland	Physical alteration and destruction of habitat, fragmentation, changes of fire regimes, invasive alien species, unsustainable logging, extraction of non-timber forest products, fuelwood extraction, hunting, unsustainable shifting cultivation, climate change, pollutants including acid rain.
Drylands	Mediterranean, grasslands, savannahs	Physical alteration and destruction of habitat, changes of fire regimes, introduced herbivores (particularly livestock), non-native plants, depletion of water resources, harvest of fuelwood, over-harvest of wild species, chemical pollution, climate change.
Agricultural	Arable land (annual crops), permanent crops, permanent pasture	Soil degradation, excessive use of fertilizer, nutrient depletion, loss of genetic diversity, loss of natural pollinators.

Source: UNDP, UNEP, WB and WRI (2000).

space allotted to hedgerows, copses, or wildlife corridors, or by displacing traditional varieties of crops with modern high-yield but uniform crops.

Ecosystem change and destruction can reduce both intra- and inter-species biodiversity. Furthermore, the increasing pressure on species through over-harvesting and hunting (of predators, for bush meat or for leisure) and the side effects of pollution processes further erode intra- and inter-species biodiversity.

The IUCN Red List published in 2006 reports that more than 16 000 species are threatened with extinction, of which 1 528 are critically endangered. Some groups of organisms are more threatened than others: the highest proportions of species threatened were for amphibians and gymnosperms (31 percent), mammals (20 percent) and birds (12 percent), while for fish and reptiles the proportion was 4 percent (IUCN, 2006).

Africa south of the Sahara, tropical South and Southeast Asia and Latin America, i.e. the regions that are home to the majority of species in the world, have a greater number of threatened species. Though alarming, the Red List figures do not represent the real scale of the problem because it was only possible to evaluate 2.5 percent of all described species (which in turn are only a small proportion of the total number of species). The difficulty of quantifying diversity of species makes the evaluation of the impacts of human activities even more difficult.

Extinction of species is a natural process, and throughout the fossil record – except for periods of mass extinction there has been a natural "background rate" of extinction. Recent extinction rates far exceed the background rates found in the fossil record. The known rate of extinctions of birds, mammals and amphibians over the past 100 years indicate that current rates are 50 to 500 times higher than background rates found in the fossil record. If "possibly extinct" species are included this increases to 100 to 1 000 times the natural extinction rates (Baillie, Hilton-Taylor and Stuart, 2004). This may be a conservative estimate, as it does not account for undocumented extinctions. Although the estimates vary greatly, current extinction rates suggest that the earth may be on the threshold of a new mass extinction event generated by human activities.

Similarly, agricultural genetic diversity is declining globally as specialization in plant and animal breeding and the harmonizing effects of globalization advance. Although 5 000 different species of plants have been used as food by humans, the majority of the world's population is now fed by less than 20 staple plant species (FAO, 2004c). And only 14 domesticated mammalian and bird species now provide 90 percent of human food supply from animals (Hoffmann and Scherf, 2006).

Forests currently host the highest number of threatened species. Many forest-dwelling large mammals, half the large primates, and nearly 9 percent of all known tree species are at some risk of extinction (WRI, 2000). The biodiversity of freshwater ecosystems is even more threatened than that of terrestrial ecosystems. Twenty percent of the world's freshwater species have become extinct, threatened, or endangered in recent decades. In the United States, which has the most comprehensive data on freshwater species, 37 percent of freshwater fish species, 67 percent of mussels, 51 percent of crayfish and 40 percent of amphibians are threatened or have become extinct (WRI, 2000). Marine biodiversity is also under great threat. Commercial species such as Atlantic cod, five species of tuna, and haddock are threatened globally, along with several species of whales, seals, and sea turtles, while invasive species are frequently reported in enclosed seas (WRI, 2000).

5.3 Livestock's role in biodiversity loss

As we have seen, the most important drivers of biodiversity loss and ecosystem service changes are habitat change, climate change, invasive alien species, overexploitation and pollution. These drivers are not independent. The impact of climate change and much of the impact of pollution on biodiversity for example is indirect, through the modification of habitats, while the latter often goes hand in hand with the introduction of invasive species.

5.3.1 Habitat change

Habitat destruction, fragmentation and degradation are considered the major category of threat to global biodiversity. They are the major threat faced by birds, amphibians and mammals, affecting over 85 percent of threatened species in all three animal classes (Baillie, Hilton-Taylor and Stuart, 2004). It has been possible to examine some of the key drivers of habitat destruction using data on birds. Large-scale agricultural activities (including crop farming, livestock ranching and perennial crops such as coffee and oil palm) are reported to impact nearly half of the globally threatened birds affected by habitat destruction. A similar proportion would be affected by smallholder or subsistence farming. Selective logging or tree-cutting and general deforestation is said to affect some 30 percent of threatened bird species, firewood collection and the harvesting of non-woody vegetation would affect 15 percent and conversion to tree plantations some 10 percent. Overall, over 70 percent of globally threatened birds are said to be impacted by agricultural activities and 60 percent by forestry activities (Baillie, Hilton-Taylor and Stuart, 2004).

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

Deforestation and forest fragmentation

Habitat change by and for livestock started from the beginning of domestication of animals, between 10 000 and 8000 BC. Around the Mediterranean Basin, clearing by fire, pastoralism and primitive agriculture were the primary impacts (Pons *et al.*, 1989). Most of the natural vegetation in the basin has since been modified by human activities. In northern temperate regions such as Europe, native vegetation has also been largely destroyed or modified by deforestation, agriculture and grazing (Heywood, 1989). In more recent times, much of the temperate forest in Australia has been converted to grassland (Mack, 1989).

Livestock production plays an important role in **habitat destruction**. At present, the link between deforestation and livestock production is strongest in Latin America, where extensive cattle grazing is expanding mostly at the expense of forest cover. By the year 2010 cattle are projected to be grazing on some 24 million hectares of Neotropical land that was forest in 2000 (Wassenaar *et al.*, 2006; see also Chapter 2). This means that about two-thirds of the deforested land is expected to be converted to pasture, with a large negative effect on biodiversity.

© JEREMY FLANAGAN

The endangered Peruvian Plantcutter Phytotoma raimondii *is endemic to the dry forest of north Peru. Conversion of forests for farming and firewood threatened the last stronghold of for the species - 2006*

In addition to pasture, a substantial and increasing share of this region's cropland, and more particularly of cropland expansion into forest, is dedicated to intensive large-scale production of soybeans and other feedcrops destined for livestock production. Between 1994 and 2004, the land area devoted to growing soybeans in Latin America more than doubled to 39 million ha, making it the largest area for a single crop, far above maize which ranks second at 28 million hectares (FAO, 2006b). The demand for feed, combined with other factors, has triggered increased production and exports of feed from countries such as Brazil where land is relatively abundant. Wassenaar *et al.* (2006) project large hotspots of deforestation in the Brazilian Amazon forest related to the expansion of cropland, mainly for soybean (see Box 5.1). Similar processes are reported to be taking place south of the Neotropics, particularly in Argentina (Viollat, Le Monde Diplomatique, April 2006).

Besides forests, this expansion of livestock-related land use has fragmented other valuable landscapes. In Brazil's ecologically very sensitive tropical savannah region, the Cerrado (recently described as the "forgotten" ecosystem – Marris, 2005), the rapid settlement and the accompanying pollution and erosion severely impact biodiversity (see Box 5.2).

It is not just the sheer area of conversion involved. The pattern that pasture expansion is taking poses a threat of **habitat degradation** through loss of neotropical biodiversity. Some 60 percent of pasture expansion into forest is projected to occur in a rather diffuse manner, in already fragmented forest landscapes (Wassenaar *et al.*, 2006). More concentrated "hotspots" of pasture expansion into forest are predominantly projected in lowland ecosystems. The tropical Andes mountain region though is the most biologically diverse of the hotspots identified by Myers *et al.* (2000), containing some

6 percent of total plant and vertebrate species world-wide. Biodiversity in the northwestern Andean moist forest and Magdaleña Valley dry and montane forest ecoregions is reported to be under severe pressure (UNEP-WCMC, 2002). These areas are projected to be affected by both pasture- and cropland-dominated diffuse deforestation.

Habitat degradation threatens many other ecoregions. Most are projected to be affected by diffuse deforestation: important examples are cropland expansion into Central American pine-oak forest and pasture expansion into the Brazilian Cerrado or the Atlantic forests of eastern Brazil which are among the most endangered habitats on earth (Myers *et al.*, 2000; UNEP, 2002). In fact almost all the diffuse deforestation areas are located in WWF's "Global 200" priority ecoregions (Olson and Dinerstein, 1998). In addition the North and Central Andes, as well as eastern coastal Brazil have extremely high densities of important bird areas (BirdLife International, 2004).

Habitat fragmentation occurs when patches of native habitat become isolated in a landscape increasingly dominated by human activities.

Under the species-area relationship it has long been recognized that large islands have more species of a given group than do small islands. For example Darlington evaluated that the reduction of an area by a factor of ten in the West Indies divides the number of species of Carabidae (beetle) by two (Darlington, 1943). Researchers are today increasingly applying this relationship to fragmented habitats and, in particular, to rain forest fragmentation stating that forest patches are hosting less biodiversity than continuous ones. In the context of forest fragmentation the decreased biodiversity would result from: a decrease in variety of habitats in the fragmented section, new opportunities for invasive alien species to intrude and compete with native ones, a decreased size of wild population easing inbreeding and eroding intra specific biodiversity, a disruption of natural equilibrium between species and in particular between prey and predators.

As a direct result, the real impacts of habitat change on biodiversity is greater when the habitat is fragmented as the actual biodiversity carrying capacity of fragmented habitats is much smaller than the overall area loss would suggest.

The effect of fragmentation on biodiversity in pasture-dominated landscapes is often aggravated by changes of fire regime. As described in Chapter 3 (Box 3.3), burning is a common practice for the establishment and management of pastures. It is practised in many grassland regions of Africa, Australia, Brazil and the United States.

Burning usually has a negative impact in large agricultural regions with fragmented natural habitat. One of the reasons is that the remaining forest fragments in these regions appear unusually vulnerable to fire, because their dryer, fire-prone edges lie adjacent to frequently burned pastures. Under the generally low prevailing level of control of burning this frequently leads to considerable penetration of fire into forest interiors (Cochrane and Laurance, 2002). Another reason is the indirect impact that fire has on biodiversity, by facilitating invasions of alien species. In a review, d'Antonio (2000) concluded that fire most often increases such invasions, even when used to control an invasive species. In addition some invasive species can also directly alter the fire regime. They can increase fire intensity in fire-prone systems or introduce fire into systems where it was previously uncommon.

Intensification of agricultural land use

In his historical perspective of biological invasions, Di Castri (1989) defines the Old World as the zone where the instruments for cultivation were the spade and particularly the plough. Deep turning of soil by ploughing has far-reaching effects on biological processes in soil, including germination. Such practices and their subsequent spread to other regions represent an

Box 5.1 The case of the protected areas

The destruction and modification of habitats in the world continues at a steady pace. According to FAO some 29.6 percent of the total land area of the world is currently under forest cover. This area is being deforested at a rate of 0.2 percent points per year (FAO, 2004).

Major efforts at the global and national levels have aimed to protect areas to safeguard key habitats and species. In 2005, 6.1 percent of the total land area of the world was under protection (WRI, 2005). This includes strict nature reserves, wilderness areas, national parks, national monuments, habitat/species management areas, and protected landscapes.

Despite the efforts to increase the number of protected areas in the world, the extinction of species and habitat losses continue. Many protected areas face significant threats including poaching, encroachment and fragmentation, logging, agriculture and grazing, alien invasive species and mining. Among those related to livestock, park managers have identified:

- incursion by nomadic groups and subsequent conflict with wild animal populations;
- establishment of ranches spreading into protected areas, and
- agricultural pollution, affecting protected areas through eutrophication and pollution by pesticides and heavy metals (Mulongoy and Chape, 2004).

Livestock pose a particular threat to protected areas.

An analysis for this report comparing global bovine density with protected areas in the top three IUCN[1] categories shows that 60 percent of the world's protected areas in these top categories have livestock (cattle and buffaloes) within a 20 km radius from the centre. Bovine density in protected areas is generally still low, but some 4 percent have an average density of four or more animals per square kilometer, representing a significant menace.

Projected land use changes in the neotropics for the year 2010 (see Maps 33A and 33B, Annex 1) show that protected areas are under further threat of livestock-linked deforestation. In Central America, for example, significant pasture expansion is expected into forest in the Maya Biosphere reserve in Guatemala's northern Petén region, mainly in the Laguna del Tigre national park. In South America, a few parks appear to be severely threatened; the Formaciones de Tepuyes natural monument in eastern Venezuelan Amazon, the Colombian national park Sierra de la Macarena and the Cuyabeno reserve in northeastern Ecuador.

Although deforestation in protected areas represents a limited portion of total deforestation, it may have a considerable ecological significance. The Macarena national park, for example, is the only remaining significant corridor between the Andes and the Amazon lowlands. Small spots of deforestation, which could be only the beginning, are also noted at the high end of the Carrasco Ichilo national park on the Andes slopes between the Bolivian highlands and the lowlands towards Santa Cruz. In all cases, the majority of the deforested area would be occupied by pasture.

[1] Category Ia or strict nature reserve: protected area managed mainly for science; Category Ib or wilderness area: protected area managed mainly for wilderness protection; and Category II or National park: protected area managed mainly for ecosystem protection and recreation.

Source: Wassenaar *et al.* (2006).

early form of intensification leading to habitat change. However, the effect on biodiversity loss has surely been far less than that resulting from intensification of agriculture through mechanization and agro-chemical use, following the industrial revolution.

In Europe today, traditional grazing is seen as having positively affected biodiversity in pastures, by creating and maintaining sward structural heterogeneity, particularly as a result of dietary choice (Rook *et al*., 2004). Other important heterogeneity-creating mechanisms are treading, which opens up regeneration niches for gap-colonizing species (although some of these may be invasive) and nutrient cycling – concentrating nutrients in patches thereby altering the competitive advantage between species. Grazing animals also have a role in propagule[3] dispersal.

However, when established traditional pastures become more intensively managed, much of the remaining diversity is lost. Today's sown pastures have lost almost all the sward canopy structure, and this effect on plant communities has led to secondary effects on invertebrate diversity, both by changing the abundance of food plants and by changing breeding sites (Rook *et al*., 2004). The direct effects of invertebrate diversity then feed through to vertebrate diversity (Vickery *et al*., 2001).

Similar effects may occur in other relatively intensive systems such as the "cut and carry" system, affecting grasslands of the more densely populated areas in developing regions, although cut and carry has considerable environmental and productivity advantages. Another aspect of more intensively managed pastures is that productivity is often hard to maintain: the export of nutrients through products and soil degradation leads to a decrease in soil fertility. This often results in increased competition among weeds and undesired grass species. The subsequent increased use of herbicides for control may constitute another threat to biodiversity (Myers and Robins, 1991).

Clearly, the recent trend towards intensive production of feedcrops, in line with the overall intensification of crop agriculture, leads to profound micro- and macro-habitat change, although the extent of the area concerned is less than for extensive pastures. Advanced technology now fosters high land-use intensity, and allows agriculture to expand into previously unused land, often in biologically valuable regions (see Box 5.2). Under such use virtually no above- or below-ground habitat remains unaffected: even within a generally very diverse soil microbial population few species may be able to adapt to the modified environment.

Desertification and woody encroachment

Another area where livestock have fuelled habitat degradation is in rangelands. Rangeland degradation results from a mismatch between livestock density and the capacity of the pasture to support grazing and trampling. Such mismanagement occurs more frequently in the less resilient arid and semi-arid regions, characterized by a relatively erratic biomass production. Section 2.5.2 describes the process in more detail. Excessive pressure on dryland ecosystems leads to fragmentation of herbaceous cover and an increase in bare soil (i.e. desertification). In semi-arid, subtropical rangelands often, though not always woody plant cover increases (Asner *et al*., 2004). Woody encroachment results when overgrazing of herbaceous cover, reduced fire frequency, helped along by atmospheric CO_2 and nitrogen enrichment, modify the equilibrium in favour of woody species.

The spread of rangeland degradation in the arid and semi-arid climates is a serious source of concern for biodiversity; although quantifying the extent is a complex exercise. Land quality indicators used to assess conditions are inadequate. There are also natural long-term

[3] Any of various, usually vegetative, portions of a plant, such as a bud or other offshoot, which also seeds, thus facilitating dispersal of the species and from which a new individual may develop.

Box 5.2 Changes in the Cerrado, Brazil's tropical savannah

The Cerrado region of woodland-savannah makes up 21 percent of Brazil's area. Large mammals such as the giant anteater, giant armadillo, jaguar and maned wolf still survive here. Biodiversity in this fragile and valuable ecosystem is endangered by a combination of fragmentation, intensification, invasions and pollution.

Like the Amazon basin, the Cerrado is a great source of biodiversity. It supports a unique array of drought- and fire- adapted plant species and surprising numbers of endemic bird species. Its 137 threatened species include the maned wolf (*Chrysocyon brachyurus*), a striking, long-legged beast that resembles a fox on stilts. The sparse, scrubby vegetation features more than 4 000 species that grow only here.

However, over the past 35 years, more than half of the Cerrado's original expanse of two million km^2 has been taken for agriculture. It is now among the world's top regions for the production of beef and soy. At the current rate of loss, the ecosystem could be gone by 2030, according to estimates by Conservation International.

Agriculture in the Cerrado started in the 1930s with extensive cattle ranching, which severely impacted the ecosystem's functioning and biodiversity. Besides altering the local vegetation by trampling and grazing, much of the impact was through damage to the neighbouring fragile natural ecosystems through fires set on pastures. The change in fire regime proved to be disastrous: the oily molasses grass (*Melinis minutiflora*), widely planted for pasture, has invaded the fringes of the wild Cerrado, causing fires to rage at such intensity that they burn through even the tough fire-adapted bark of native woody plants.

Still, the Cerrado's inaccessibility and poor soil spared large areas from large-scale exploitation. As Brazil embraced the Green Revolution in the 1970s, the availability of new soy varieties and fertilizers turned the region into a viable agricultural prospect. Soybean cultivation has since invaded the Cerrado where national production increased by 85 percent between 1993 and 2002. Soybean production in the Cerrado is characterized by high intensity land management, known as the "Patronal" model, based on advanced technology, full mechanization and extensive use of agrochemicals. Production units are generally well over 1 000 ha. This intensive system allows for high productivity: soy is harvested twice a year sometimes with an intermittent maize crop.

The replacement of originally rich habitats by an intensive monoculture landscape strongly affects biodiversity. Habitats have been lost on a large scale and pesticides and fertilizers, sprayed in large quantities to control pests and diseases and to maintain fertility, pollute the water and the soil. Though the use of herbicides against weeds is on the increase, weeds were previously dealt with using mechanical methods that have favoured erosion; WWF (2003) estimates that a soy field in the Cerrado loses approximately 8 tonnes of soil per hectare every year.

There is a growing realization among conservationists that their strategies must accommodate economic development (Odling-Smee, 2005). To this end, ecologists working in the Cerrado are now stressing the ecosystem services it provides — many of which have a tangible economic value. Some are investigating the role of the native landscape as a carbon sink, as a centre of genetic diversity for the crop cassava, or as a protector of Brazil's soil and water.

Source: Marris (2005).

Le Bheyr lake is of vital importance to the microclimate of the zone. Apart from providing grazing along its shores, it is a fishing and crossing point for migratory birds in December and January. The photo shows striking images of environmental degradation and drought – Mauritania 1996

oscillations in ecosystem changes that are difficult to disentangle from anthropogenic changes. However, many grazing systems are undergoing desertification. Africa, Australia and the southwestern United States have experienced a severe reduction in plant populations, with a corresponding loss of biodiversity. Often they are dominated by one or a few woody species, with little herbaceous canopy remaining (see review by Asner *et al.*, 2004). Biodiversity erosion creates a negative feedback: it reduces the system's resilience and thereby indirectly reinforces desertification. This acknowledged inter-linkage has led to the development of a joint work programme between the United Nations Convention to Combat Desertification (UNCCD) and the Convention on Biological Diversity (CBD).

Vegetation-grazing interactions associated with woody encroachment strongly depend on grazing intensity. Grazing probably facilitates bush encroachment, and thus system structure, by reducing risk of fire for woody seedlings. Grazing also encourages erosion on some landscapes, which affects the herbaceous cover more than the deeper-rooted vegetation. Reduction of herbaceous cover through grazing can also advantage woody vegetation in the competition for access to limited resources such as water. Changes are more pronounced in cases of long-term, heavy grazing (see the example of Texas in Box 5.3). Woody encroachment sometimes results from concentration of grazing pressure that has occurred because of declines in the mobility of pastoral people and their herds. Under heavy grazing, herbaceous cover is often replaced by woody vegetation while perennial grasses replace annual ones.

Effects of woody species on the herbaceous community vary according to the type of woody species and site. Effects can be positive, neutral or negative. The change from grassland to woodland through the process of woody encroachment affects several key ecosystem functions, including decomposition and nutrient cycling, biomass production and soil and water conservation. The dynamics of rainfall interception, overland flow and water penetration into the soil in overgrazed areas often is such that water from rainfall events is quickly lost to drainage systems with a concomitant increase in soil erosion. Pristine grassland may intercept water more efficiently and, therefore, prevent loss of the soil resources that form the basis of the entire ecological and agricultural production system. In arid environments, effects are eventually mostly negative both for animal production and biodiversity. Habitat diversity may also be affected. Savannah-like openings in wooded landscapes for example may gradually vanish as a result of woody encroachment.

Forest transition and the conservation of pastoral landscapes

Forest transition, i.e. the process of previous agricultural land being turned back into forest - was presented in Section 2.1.2. This increasingly widespread land use change process is characterized by the abandonment of agricultural land in remote areas with poor soil. These are predominantly pastures, which when abandoned, can regenerate back into forest.

Box 5.3 Woody encroachment in southern Texas

The woody plants that invade areas during woody encroachment are typically species that were present somewhere in the landscape before the introduction of grazing. For example, in a southern Texas rangeland containing a diverse array of trees, shrubs, and subshrubs, heavy grazing caused increases in the cover of the nitrogen-fixing tree *Prosopis glandulosa var. glandulosa* (mesquite). Long-term records and aerial photographs indicate that mesquite encroachment then facilitated the establishment of other woody plants in its understory, which subsequently out-competed mesquite for light and other resources. Mesquite remnants are commonly found among well-developed patches of woody vegetation known not to have existed a century ago.

Source: Extract from Asner *et al*. (2004).

Some abandoned pastures turn into fallow/shrubland with little biological diversity. In temperate regions such as Europe, natural and semi-natural grasslands have become an important biodiversity and landscape resource worth preserving in their own right. These plant communities, and the landscapes of which they form a part, are now highly valued and the subject of numerous agro-environmental and nature conservation schemes. These habitats are under threat from two contrasting directions: on the one hand, the ongoing intensification of land use, and on the other, an increasing number of former meadows and pastures lying fallow owing to changing economic conditions and "set-aside" subsidies.

As early as 1992, Annex 1 of the European Council Habitat Directive (EU, 1992, cited in Rook *et al*., 2004) listed habitats that are considered of European importance for their biodiversity value. It has been estimated that this list includes 65 types of pasture habitat that are under threat from intensification of grazing and 26 that are under threat from abandonment (Ostermann, 1998). In some cases, there is not only a loss of biodiversity value but also other environmental problems. For example, in the hills and mountains of Mediterranean countries there are now large areas of former grazing covered by shrub vegetation of very low biodiversity. This accumulation of woody biomass may increase risks such as fire and erosion, resulting in extensive environmental and economic losses (Osoro *et al*., 1999).

One of the main objectives of nature conservation in Europe is, therefore, to protect semi-open landscapes. In several countries the establishment of larger "pasture landscapes" with a mixed character of open grassland combined with shrubs and forests has been recognized as one solution (Redecker *et al*., 2002).

Within grassland communities spatial heterogeneity is the key to maintaining critical biodiversity. The role of the grazing animal in fostering this has already briefly been mentioned under "Intensification of agricultural land use," above.

Woodland pastures (Pott, 1998; Vera, 2000) harbour higher biodiversity as they contain both grassland and forest species. A different mix of grazers and browsers may be needed to manage such landscapes (Rook *et al*., 2004). In premodern times, woodland pastures were used for communal grazing: today the challenge is to develop analogous grazing systems that achieve similar biodiversity but are socio-economically viable. Vera (2000) argues that long-term preservation of biodiversity requires the development of wilderness areas with wild herbivores in addition to the existing semi-natural landscapes.

Examples of species extinction at least partly resulting from livestock induced habitat change

A few positive roles of livestock have been mentioned with respect to habitat change, concerning either its role in habitat regeneration or in maintaining a relatively slow pace or low level of change (see also Sections 5.3.4 and 5.5).

Still, it is clear that while not all indirect effects have been analysed, other aspects of livestock production have affected many habitats badly at enormous scales. The table on livestock's contribution to species extinction via habitat loss or habitat degradation (Table 16, Annex 2) gives specific examples of how these various mechanisms have led to the loss of particular species. It shows clearly that habitat degradation by and for the livestock sector has contributed to the extinction of many plants and animals. Nevertheless, it is unknown what the status of the affected habitats would have been in the absence of livestock.

5.3.2 Climate change

The impact of climate change on biodiversity is recent, and only now starting to be recognized, observed on the ground and understood. Climate change affects biodiversity in three main ways: by changes in the mean climate, changes in the incidence or severity of extreme climate events and changes in climate variability.

According to Thomas *et al*. (2004) between 15 and 37 percent of all species could be threatened with extinction as a result of climate change.

The projected impacts on biodiversity owing to climate change include the following (Secretariat of the Convention of Biological Diversity, Technical Series No. 10, 2003):

- As a result of global warming, the climate range of many species will move poleward or upward in elevation from their current locations. Species will be affected differently by climate change: some will migrate through fragmented landscapes whilst others, less mobile, may not be able to do so.
- Many, already vulnerable species are likely to become extinct, especially species with limited climate ranges and/or with limited geographical opportunities (e.g., mountain top species, species on islands, peninsulas). Species with restricted habitat requirements, very large ranges, slow breeding rates or small populations are typically the most vulnerable.
- Changes in the frequency, intensity, extent and locations of climatically- (and non-climatically-) induced disturbances will affect how existing ecosystems will be replaced by new plant and animal assemblages. Species are unlikely to migrate at the same rates; long-lived species will persist longer in their original habitats leading to new plant and animal assemblages. Many ecosystems will become dominated by opportunistic, 'weedy' species, well adapted to dispersal and rapid establishment, especially if the frequency and intensity of disturbance is high.
- Some ecosystems are particularly vulnerable to climate change, such as coral reefs, mangroves, high mountain ecosystems, remnant native grasslands and ecosystems overlying permafrost. Some ecosystems may be slow to show evidence of change, whilst others, e.g. coral reefs, are already showing a rapid response. The net primary productivity (NPP) of many plant species (including some but not all crop species) increase due to the "fertilizer effect" of rising concentrations of atmospheric carbon dioxide. However, when temperature, nutrient limitation and rainfall changes are also considered there may be losses in net ecosystem and biome productivity in some regions. The differential changes in NPP will result in changes in the composition and functioning of ecosystems. Losses in net ecosystem and biome productivity can occur, for example, in some forests, at least when significant ecosystem disruption occurs (e.g. loss of a dominant species or a high proportion of species owing to changes in incidence of disturbances such as wildfires, pest and disease outbreaks).

Many studies suggest that climate change (including its effects on habitats) will surpass other, more direct, forms of human-induced habitat change as the main threat to biodiversity loss. In any case, the combined impact of continued habitat loss and climate change will pose a major and potentially catastrophic threat to biodiversity in the future. The changes to current pristine areas resulting from climate change will force species to move to and through already degraded and fragmented habitats, worsening their opportunities of dispersal and their chances of survival.

The IPCC (2002) has reviewed the extent to which biodiversity has already begun to be affected by climate change. Higher regional temperatures have affected the timing of reproduction in animals and plants and/or migration of animals, the length of the growing season, species distributions and population sizes, and the frequency of pest and disease outbreaks.

The IPCC modelled the impact of four different climate change scenarios on biodiversity, producing impact scenarios for different world regions. Climate change is projected to affect individual organisms, populations, species distribution and ecosystem function and composition both directly through heat, drought, and indirectly through changes in the intensity and frequency of disturbances such as wildfires. The IPCC observes that a realistic projection of the future state of the earth's ecosystems would need to take into account human land- and water-use patterns, which will greatly affect the ability of organisms to respond to climate change. Many other information needs and assessment gaps persist, partly because of the extreme complexity of the issue.

What is livestock's contribution to the loss of biodiversity induced by climate change? Since climate change is a global process, livestock's contribution to the resulting erosion of biodiversity is in line with its contribution to climate change (see Chapter 3 for a detailed assessment). As a major driver behind landscape and habitat changes, the livestock sector may also aggravate the impact of climate change on biodiversity, by making it more difficult for climatically-challenged organisms and species to migrate across fragmented and disturbed habitats and human agricultural and urban environments. However, a shift to well-managed industrial intensive livestock production systems, by reducing the area taken up by livestock production, may work to reduce this effect.

5.3.3 Invasive alien species

Before modern times, natural ecosystems evolved in isolation on the various continents and large islands, constrained by biogeographic barriers such as oceans. Today, almost all these ecosystems have become functionally connected by the human capacity to transport biological material long distances in a short amount of time. Humans have transported animals and plants from one part of the world to another for thousands of years, sometimes deliberately (for example livestock released by sailors onto islands as a source of food) and sometimes accidentally (e.g. rats escaping from boats). Many of the world's major crops were deliberately transplanted from one continent to another – for example, maize, potatoes, tomatoes, cocoa and rubber from the Americas to the rest of the world. Following human-assisted introduction, many alien species became invasive, i.e. their establishment and propagation led to ecological and/or economic harm.

Invasive species can affect native species directly by eating them competing with them, and introducing pathogens or parasites that sicken or kill them or, indirectly, by destroying or degrading their habitat. Invasive alien species have altered evolutionary trajectories and disrupted many community and ecosystem processes. In addition, they can cause substantial economic losses and threaten human health and welfare. Today invasive species constitute a major threat affecting 30 percent of globally threatened birds, 11 percent of threatened amphibians and 8 percent of the 760 threatened mammals for which data are available (Baillie, Hilton-Taylor and Stuart, 2004).

The contribution of the livestock sector to detrimental invasions in ecosystems goes well beyond the impact of escaped feral animals.

Because of the many forms this contribution takes, the overall impact in this category of threat is perhaps even too complex for accurate assessment. One such other dimension is livestock's role as an important driver behind habitat change leading to invasions. Animal production has also sometimes driven intentional plant invasions (for example, to improve pastures). On a different scale grazing animals themselves directly produce habitat change facilitating invasions. Movement of animals and animal products also makes them important vectors of invasive species. Livestock have also been a victim of alien plant species invasions in degrading pasture land, which may in turn have driven pasture expansion into new territories. We will examine these different dimensions in the rest of this section.

Livestock as an invasive species

According to IUCN (2000) an invasive alien species is one that becomes established in natural or semi-natural ecosystems or habitats and threatens native biological diversity. Under this definition livestock can be considered as alien species that are invasive, particularly when little attempt is made to minimize the impact on their new environment, leading to competition with wildlife for water and grazing, the introduction of animal diseases and feeding on seedlings of local vegetation (feral animals are among the main threats to biodiversity on islands). The IUCN/SSC Invasive Species Specialist Group (ISSG) classifies feral cattle, goats, sheep, pig, rabbits and donkeys as invasive alien species (among a total of 22 invasive mammalian species)[4]. Indeed, feral pigs, goats and rabbits are classified among the top 100 world's worst invasive alien species.

One of the best documented effects of invasive-species is the dramatic impact of mammalian herbivores, especially feral goats and pigs, on the vegetation of small islands, causing extinction of native species and pronounced changes in dominance and physiognomy and directly affecting many other organisms (Brown, 1989). As invasive alien species, feral animals also contribute to biodiversity loss at the continental level. Nearly all livestock species of economic importance are not native to the Americas, but were introduced by European colonists to the Americas in the sixteenth century. Many harmful feral populations resulted from these introductions and the often very extensive patterns of management.

Despite the negative impact of some introduced species, exotic vertebrates continue to be imported. Government agencies are gradually becoming more cautious, but they continue deliberately to introduce species for fishing, hunting and biological control. The pet trade is perhaps the single largest source of current introductions (Brown, 1989). The contribution of the livestock sector to current vertebrate introductions is currently minimal.

Other direct livestock contributions remain important. Seed dispersal by vertebrates is responsible for the success of many invaders in disturbed as well as undisturbed habitats. In Australia, more than 50 percent of naturalized plant species are dispersed by vertebrates (Rejmánek *et al.*, 2005). Grazing livestock have undoubtedly contributed substantially to seed dispersal and continue to do so. However, seed dispersal by vertebrates is a complicated process; when and where vertebrates promote plant invasions requires substantially more research (Rejmánek *et al.*, 2005).

Dispersal by trade in animal products is also poorly documented. An interesting exception is the detailed analysis of the impact of the increased demand for wool in the early twentieth century. The monograph of Thellung (1912) on the adventive flora of Montpellier was largely inspired by the expansion of alien species resulting from the import, hanging out and drying of wool at Port-Juvénal (near Montpellier). It is not

[4] http://issg.appfa.auckland.ac.nz/database/welcome/

Box 5.4 Wild birds and highly pathogenic avian influenza

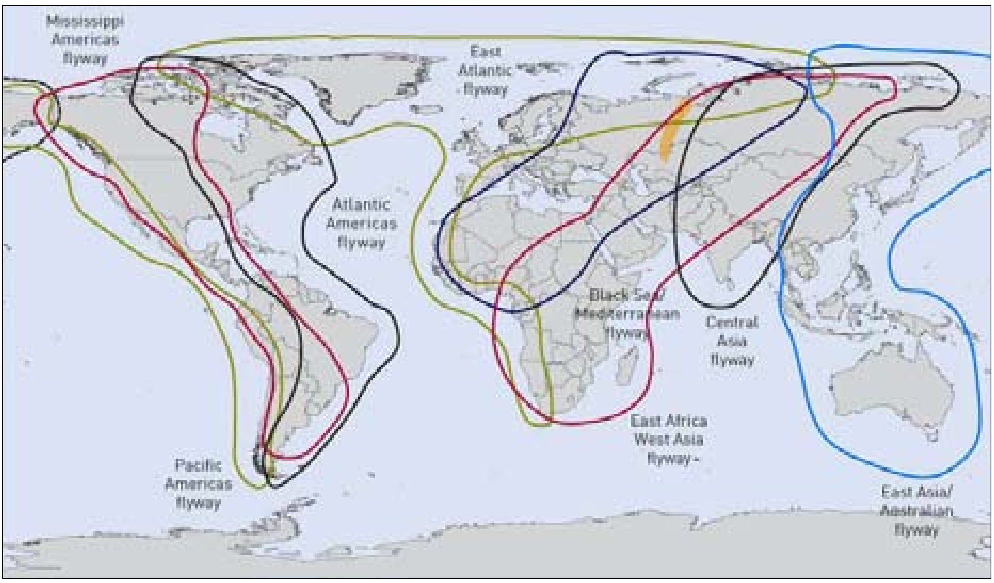
There is a possible and plausible link between wild birds and poultry in the transmission of highly pathogenic avian influenza (HPAI) that has recently affected the poultry sector worldwide and raised concern over human health. Since 2003, there has been a series of outbreaks of this new disease. By July 2006, the disease had affected the poultry industries in 55 countries; 209 million birds were killed by the disease or had to be culled. HPAI is a zoonotic disease, which is potentially fatal to human beings. By July 2006, disease had been caused in 231 cases, killing 133 people. The disease has now become endemic in several countries in Asia and Africa.

The widespread simultaneous occurrence of the disease poses a substantial risk of a potential disruption to the global poultry sector (McLeod *et al.*, 2005). The emergence of the specific strain of HPAI involved in these recent outbreaks, called H5N1, raises concerns regarding the potential role of wild birds as one possible transmission mechanism (Hagemeijer and Mundkur, 2006).

Before the Asian H5N1 epidemic in 2003, HPAI was considered a disease of domestic birds. Wild aquatic birds of the world were only known as natural reservoirs of low pathogenic influenza A. The series of initial outbreaks, particularly in Asia has pointed to possible interactions between domesticated and wild bird populations in HPAI virus transmission (Cattoli and Capua, 2006; Webster *et al.*, 2006).

Bird migratory patterns annually connecting land masses from the northern and southern hemispheres (including the African-Eurasian, Central Asian, East Asian- Australasian and American flyways) may contribute to the introduction and to

Map 5.1 Major flyways of migratory birds (Shore birds)



Source: Flyways – Wetlands International.

Box 5.4 **cont.**

the spread of the infection to AI-free areas. Recent outbreaks of HPAI in Africa, Central Asia, Europe and the Russian Federation suggest that A/H5N1 may have been carried by wild birds during their autumn and spring migrations (Cattoli and Capua, 2006; Hagemeijer and Mundkur, 2006). In particular, migratory wild birds were found positive in many European countries with no associated outbreaks in poultry (Brown *et al.*, 2006).

On the other hand wild bird populations could possibly be contaminated and impacted by infected poultry units. According to Brown *et al.* (2006) further infection of wild birds through exposure to infected 'backyard' poultry in Eastern Europe appears probable.

known whether today's much stricter sanitary regulations impede the sharply increasing global trade in animal products from having similar impacts.

Historically, livestock played an important role in the transmission of disease organisms to populations that had no immunity. The introduction of rinderpest into Africa at the end of the nineteenth century devastated not only cattle but also native ungulates. This transmission remains an issue in today's world. The introduction of avian pox and malaria into Hawaii from Asia has contributed to the demise of lowland native bird species (Simberloff, 1996).

Even if there is no sound evidence as yet of cross-contamination between wild and domesticated bird populations, this mechanism possibly plays a role in today's spread of highly pathogenic avian influenza (see Box 5.4).

Livestock-related plant invasions

The natural temperate grasslands of Australia, South America and western North America offer some of the most extreme examples of what has been called "the great historical convulsions" of the earth's biota – massive changes in the species composition of once vast communities through the transoceanic transport of alien organisms and their subsequent incursion into new ranges (Mack, 1989). In less than 300 years (and mostly only some 100 years) much of the temperate grassland outside Eurasia has been irrevocably transformed by human settlement and the concomitant introduction of alien plants.

Clearly, livestock production was only one among many other activities driving the largely unintentional trans-Atlantic movement of alien species. However, large ruminants are considered to have largely enhanced the invasive potential of these species. According to Mack (1989), the two quintessential characteristics that make temperate grasslands in the New World vulnerable to plant invasions are the lack of large, hooved, congregating mammals[5] in the Holocene or earlier, and the dominance by caespitose grasses (which grow in tussocks). The morphology and phenology of such grasses make them vulnerable to livestock-facilitated plant invasions: the apical meristem becomes elevated when growth is resumed and is placed in jeopardy throughout its growing season to removal by grazers, while these grasses persist on site exclusively through sexual reproduction. In caespitose grasslands trampling can alter plant community composition by destroying the

[5] The only exception are enormous herds of bison that were supported on the Great Plains of North America, yet these large congregating animals occurred only in small, isolated areas in the intermountain west. The phenology of caespitose grasses may account for this paucity of bison (Mack, 1989). In both vulnerable grasslands in western North America the native grasses on zonal soils are all vegetatively dormant by early summer when lactating bison need maximum green forage.

matrix of small plants between the tussocks.

Once European settlers arrived, alien plants began to colonize these new and renewable sites of disturbance. Whether through grazing or trampling, or both, the common consequence of the introduction of livestock in the three vulnerable grasslands were the destruction of the native caespitose grasses, dispersal of alien plants in fur or faeces, and continual preparation of a seed bed for alien plants. Even today, New World temperate grasslands are probably not yet in a steady state, but are certain to experience further consequences from existing and new plant invasions (Mack, 1989).

Besides the natural grasslands, the world's managed pastures owe their origin and history to human action. Livestock-related land-use changes continue, as do their impacts on biodiversity through habitat destruction and fragmentation. These areas are often rich in alien invaders, some of them deliberately introduced. Planned invasions have taken place in vast areas of tropical savannah, often assisted by fire. Such invasions have a long history in Australia as reviewed by Mott (1986). With the exception of some savannahs of edaphic origin, the grassland ecosystems in Africa usually result from the destruction of forest or woodland. They are often maintained through the use of fire regimes and are frequently invaded by alien species (Heywood, 1989). Likewise, in South America, the region of the great savannahs, including the cerrados and campos of Brazil, and the llanos of Colombia and Brazil have become increasingly exploited leading to invasion by weedy and pioneer species. Many of the ranch lands of South America were established on previous forest land after the European-led colonization. Similarly, in Madagascar vast areas of the natural vegetation have been burned since Palaeo-Indonesians invaded the island, to provide pasture land for zebu cattle, and are burned annually. These pastures are now largely devoid of trees and shrubs and low in biodiversity and characterized by weedy species (Heywood, 1989).

Invasive species threats to pasture

Some invasive alien species alter grazing lands in a detrimental way. These include many thistle species found on most continents (see the case of Argentina in Box 5.5). In California, Star Thistle was introduced during the gold rush as a contaminant of alfalfa. By 1960 it had spread to half a million hectares, to 3 million hectares by 1985, and nearly 6 million by 1999 (Mooney, 2005). It alters the ecological balance, particularly through depletion of water, and degrades pasture value. According to Gerlach (2004) it causes soil moisture losses that represent 15 to 25 percent of mean annual precipitation, representing a value of lost water ranging between US$16 and 75 million per year in the Sacramento River watershed alone. Together with other invasive weeds such as Black Mustard it causes more than US$2 billion of damage annually (Di Tomaso, 2000). A grass that is widespread and used for permanent pastures in various parts of the tropics is *Axonopus affinis*. It invades degenerated pastures of *Paspalum dilatatum, Trifolium repens* and *Pennisetum clandestinum*, leading to a decline in animal production (UNESCO, 1979). Major problems are caused by other introductions such as *Lantana camara*, one of the world's ten worst weeds (GISD, 2006), which has invaded many natural and agricultural ecosystems of the Palaeotropics. The replacement of native pastures by Lantana is threatening the habitat of the sable antelope in Kenya and Lantana can greatly alter fire regimes in natural systems. It is toxic to livestock (in some countries, it is therefore planted as a hedge to contain or keep out livestock). At the same time it benefits from the destructive foraging activities of introduced vertebrates such as pigs, cattle, goats, horses and sheep creating micro habitats for germination. It has been the focus of biological control attempts for a century, yet still poses major problems in many regions.

Box 5.5 **From pampas to cardoon, to alfalfa, to soy**

The Pampas, the humid grasslands of northern Argentina dominated by caespitose species, were the site of one of the earliest documented and dramatic transformations of a landscape by alien plants. In the Origin of Species (1872) Darwin remarked that the European cardoon (*Cynara cardunculus*) and a tall thistle (*Silybum marianum*) "are now the commonest [plants] over the whole plains of La Plata, clothing square leagues of surface almost to the exclusion of every other plant." Even in Southern Uruguay he found "very many square miles covered by one mass of these prickly plants, impenetrable by man or beast. Over the undulating plains, where these great beds occur, nothing else can now live." These scenes had probably arisen in less than 75 years.

Von Tschudi (1868) assumed that the cardoon arrived in Argentina in the hide of a donkey. Many early plant immigrants probably arrived with livestock, and for 250 years these flat plains were grazed but not extensively ploughed (Mack, 1989). Cardoon and thistle were eventually controlled only with the extensive ploughing of the pampas at the end of the nineteenth century.

However, this was far from the end of livestock-related plant invasions. The transformation of the pampas from pasture to farmland was driven by immigrant farmers, who were encouraged to raise alfalfa as a means of raising even more livestock. This transformation greatly expanded the opportunity for alien plant entry and establishment. Towards the end of the nineteenth century over 100 vascular plants were listed as adventive near Buenos Aires and in Patagonia, many of which are common contaminants of seed lots. More recent "immigrant" species pose further threats in the pampas and Patagonia. Marzocca (1984) lists several dozen aliens officially considered as "plagues of agriculture" in Argentina.

While the massive transformation of Argentinean vegetation continues, the globalizing livestock sector recently drove yet another revolution of the pampas. In just a few years, soybean has become the country's major crop. In 1996 a genetically modified soybean variety entered the Argentinean market with a gene that allowed it to resist herbicides. Other important factors contributed to the success of what is now called "green gold". The extensive erosion of the Pampa soils (the GM soybean is cultivated without tillage, which reduces erosion), the sharp increase in demand since the European mad cow crisis and the devaluation of the Argentinean peso. Upon arrival of the GM variety in 1996, soybean covered six million hectares, while today it covers 15.2 million hectares, i.e. more than half Argentina's arable land. Rates of deforestation now exceed the effect of previous waves of agricultural expansion (the so-called cotton and sugarcane "fevers") (Viollat, 2006). At the same time the intensive cropping of soybean results in a severe mining of soil fertility. Altieri and Pengue (2006) estimated that in 2003 soybean cropping extracted a million tonnes of nitrogen and some 227 000 tonnes of phosphorus, losses that would cost some US$910 million if replaced by mineral fertilizers.

Cardoon (Cynara cardunculus) in Shoreline Park, Mountain View, California – United States 2003

Sources: Mack (1989) and Viollat (2006).

Feed-crop related threats to biodiversity

Even the biodiversity of the world's cultivated crops is under threat because the narrowing genetic base of many of the world's crops put them at risk. This concern is reflected in the International Treaty on Plant Genetic Resources for Food and Agriculture, adopted by the member states of FAO in 2001. Important feedcrops like sorghum and maize are among the priority crops. Much of the genetic erosion of such staple crops occurred as a consequence of the Green Revolution, while currently there is substantial controversy around the effects to be expected from modern genetic engineering. Evidence is insufficient, but there exists strong societal concern about the possible contamination of conventional varieties by genetically modified ones, a mechanism that could be considered as "invasion". A much cited case is the contamination of local maize varieties in Mexico, the world's original centre of maize diversity, by commercial trans-genetic varieties cultivated for feed in the United States (Quist and Chapela, 2001), although this has been challenged (Marris, 2005). Similar concern exists for soybean, mainly cultivated for feed, because in countries such as the United States and Argentina (Box 5.5) genetically modified varieties tend to largely substitute conventional varieties.

5.3.4 Overexploitation and competition

Overexploitation refers to the unsustainable use of species for food, medicine, fuel, material use (especially timber), and for cultural, scientific and leisure activities. Over-exploitation has been identified as a major threat affecting 30 percent of globally threatened birds, 6 percent of amphibians, and 33 percent of evaluated mammals. It is believed that when mammals are fully evaluated for threats, overexploitation will prove to affect an even higher percentage of species (Baillie, Hilton-Taylor and Stuart, 2004). Among mammals threatened by over-exploitation, larger mammals, especially ungulates and carnivores, are particularly at risk. Mammals are used extensively in the wild meat trade, notably in tropical Africa and in Southeast Asia. Some mammal species are also harvested for medicinal use, especially in eastern Asia. Overexploitation is seen as the leading threat to the world's marine fishes.

The livestock sector affects overexploitation of biodiversity mainly through three distinct processes. Competition with wildlife is the oldest and renown problem, which often leads to reduction of wildlife populations. More recent processes include overexploitation of living resources (mainly fish) for use in animal feed; and erosion of livestock diversity itself through intensification and focus on fewer, more profitable breeds.

Competition with wildlife

Herder-wildlife conflicts

Conflicts between herders and wildlife have existed since the origins of livestock domestication. The competition arises from two aspects: direct interactions between wild and domesticated animal populations and competition over feed and water resources.

During the origins of the domestication process the main threat perceived by herders was predation by large carnivores. This led to large carnivore eradication campaigns in several regions of the world. In Europe, this led to the local extinction of several species including

© FAO/17043/G. BIZZARRI

Wild elephants and cattle competing for natural resources – Sri Lanka 1994 wolves and bears. In Africa, these tensions have led to a constant pressure on lion, cheetah, leopard and African wild dog populations.

Conflicts between herders and predators still persist in regions where extensive production systems are predominant and where carnivore populations still exist or have been reintroduced. This is the case even in developed countries, even though the predation pressure is lower and herders are usually compensated for their losses. In France, for example, the reintroduction of the wolf and the bear in the Alps and Pyrenees has led to intense conflicts between pastoral communities, environmental lobbies and the government.

In developing countries the conflicts can be acute. In sub-Saharan Africa, especially in East and Southern Africa, production losses from predation can be an economic burden to local communities. In Kenya these losses can represent up to 3 percent of the annual economic value of the herd: it is estimated that a single lion costs the herder community between US$290 and US$360 per year in production losses. Annual losses amount to US$15 for an African wild dog, US$211 for a leopard, US$110 for a cheetah, and US$35 for a hyena (Frank, Woodroffe and Ogada, in press; Patterson *et al.*, 2004; Woodroffe *et al.*, 2005). These losses compare to gross domestic product per capita of US$320 per year in Kenya. Even if the national economic impact remains negligible, the local and individual impact can be dramatic, particularly for poor people (Binot, Castel and Canon, 2006).

Predation pressure, and negative attitudes to predators among local populations, is worsening in the surroundings of the National Parks in developing countries, especially in East Africa. On the one hand, many of the protected areas are too small to host viable populations of large carnivores, as these populations often need vast hunting territories and so are forced to range outside of the parks. For example, the African wild dog in Africa has a hunting territory that extends over 3 500 km^2 (Woodroffe *et al.*, 2005). On the other hand, as land pressure mounts and traditional rangelands are progressively encroached by cropping, herders are often forced to graze their animals in the direct vicinity of the national parks. During dry seasons the surroundings of the national parks which are rich in water and palatable fodder, are often very attractive to the herders. There are, therefore, close contacts between wild predators and livestock.

Another source of intensifying conflict is that, as populations of wild ungulates are shrinking, wild predators are forced to look for other prey. Livestock do not represent a food of preference for the large carnivores, but they are easily accessible and large carnivores can get used to them. Conflicts between wild predators and livestock are, therefore, becoming frequent and acute (Frank, Woodroffe and Ogada, 2006; Patterson *et al.*, 2004; Binot, Castel and Caron, 2006).

The perception that wildlife is a threat to livestock has evolved considerably during the twentieth century. With a better understanding of the dynamics of infectious disease, herbivores, omnivores and bird populations came to be seen as disease reservoirs (buffaloes for cattle, boar for pig), as disease vectors, or as intermediary hosts (arthropod vectors such as tsetse fly for trypanosomiasis, molluscs such as *Lymnaea spp.* for the liver fluke *Fasciola hepatica*). Measures to limit the transmission of pathogens and parasites included the massive eradication of the vectors, and the limitation of contacts between the wild and domesticated animal population. In some cases, the eradication of wild mammalian species has been considered where they are disease reservoirs (the badger in Great Britain is considered a potential reservoir of tuberculosis for cattle) (Black, 2006). This threat has been exacerbated by the fact that it applies to both extensive and intensive production systems, where the introduction of new pathogens can have a dramatic impact (as suspected for avian influenza).

This wildlife-livestock interface is of acute importance to the livestock sector. It used to be

Herd of cattle entering reserve where forage is guaranteed for the animals – Mauritania 1996

an issue of local or regional dimensions (rinderpest in Africa). It has now become a global threat as demonstrated by the current avian influenza pandemic where wild bird populations may have a role in disease transmission.

Protected areas at risk of encroachment

Besides the direct interactions between wildlife and livestock resulting from predation and disease transmission, extensive livestock systems are increasingly competing with wildlife for access to land and natural resources in the African rangelands. Extensive production systems and wildlife have intermingled together for millennia in the dry lands of Africa, making simultaneous use of common resources. The actors' two forms of land use were compatible as pastoralism used natural resources with minimal impact in connection to land management and transformation. Furthermore, because of the high mobility of extensive production systems in Africa, their impact on resources was negligible and competition over access to common resources was low (Bourgeot and Guillaume, 1986; Binot, Castel and Canon, 2006).

Another form of competition for land between livestock and wildlife is the spread of protected areas. In the twentieth century most of the protected areas were created at a time when land was abundant and opportunity cost for the local communities was low. Nevertheless, with the extension of National Parks, and the spread of crop farming, extensive production systems were progressively deprived of an important part of their potential resources increasing the risk of potential conflicts. Today, protected and hunting areas represent almost 13 percent of the land in sub-Saharan Africa (Roulet, 2004). Under current population and land-use trends, the opportunity costs associated with protected areas are increasing, and are especially high in times of drought or conflict. The surroundings of these areas are under great pressure as they are often rich in water and fodder resources compared to the other, often degraded lands available. The interactions between wild fauna and livestock production systems is often localized on the peripheries of these conservation areas (Ballan, 2003; Rodary and Castellanet, 2003; Benoît, 1998; Convers, 2002).

Mobile herders often have great difficulties understanding the logic behind conservationist activities, especially when their cattle are threatened by thirst and famine while resources remain plentiful for the wild animals. To save their herds, or to minimize the conflicts with the croppers, herders are often tempted to graze their animals in the national parks. These actions have usually led to dramatic repression in the past, and herds grazing within protected areas have sometimes been slaughtered. Intense repression around parks has worsened the conflicts between conservation objectives and local communities (Toutain, 2001; Barraud, Salen and Mamis, 2001).

This situation was also worsened by policies that ignored the importance of mobility in extensive production systems in dry lands with their highly variable and shifting local rainfall, and the potential complementarities between conservation and pastoralist needs in terms of mobility. In Africa, policies encouraging settlement or sedentarization of pastoral nomads often included fencing to demarcate newly-created ranches. Nevertheless, as has been observed around Nairobi National Park, as soon as the first drought depleted ranch resources, herders decided to leave the ranches in search of water and green pasture. Often the land was sold to newcomers for cropping activities and fragmented into smaller plots. As more land is fenced, migratory routes for wildlife and nomads are blocked and both systems are impacted, increasing the risk of further conflicts (Binot, Castel and Caron, 2006).

One approach to reducing the conflicts between wildlife and livestock in the rangelands consists of working on the land-use complementarities between the two actors. This approach is, nevertheless, often opposed by conservation and livestock development programmes, as it may favour the transmission of diseases and may increase poaching pressure if regulatory mechanisms fail (Binot, Castel and Caron, 2006).

Overfishing

The role of fishmeal as a livestock feed

An important contribution of livestock to overexploitation consists in the production of fishmeal for livestock feed.

The world's ocean fish face serious threats to their biodiversity. The principle source of pressure is overexploitation by fisheries, which have affected the size and viability of fish populations, the genetics of target species, and the food chains and ecosystems of which they are part. FAO (2005b) estimates that 52 percent of the world stocks are fully exploited, and are therefore producing catches that are already at or very close to their maximum sustainable production limit, with no room for further expansion, and even some risk of decline if not properly managed. Approximately 17 percent are overexploited and 7 percent depleted.

The stocks of seven of the top ten species, accounting for 30 percent of the world total marine capture fisheries production, are either fully exploited or overexploited and, therefore, no sustainable increases in catches can be expected from these species. These include two stocks of Peruvian anchoveta (*Engraulis ringens*, an industrial "feed-grade" fish, which according to the International Fishmeal and Fish Oil Organization) are overexploited in the southeast Pacific after recovering from a recent decline; Alaska Pollock (*Theragra chalcogramma*), fully exploited in the North Pacific; Japanese anchovy (*Engraulis japonicus*), fully exploited in the northwest Pacific; blue whiting (*Micromesistius poutassou*), overexploited in the northeast Atlantic; capelin (*Mallotus villosus*) fully exploited in the North Atlantic; and Atlantic herring (*Clupea harengus*) with several stocks in the North Atlantic, most of them fully exploited. The latter three are largely used to produce fishmeal (Shepherd *et al*., 2005). The Chilean jack mackerel, another important fishmeal species, is assessed as fully or overexploited and yielded 1.7 million tonnes in 2002, having declined continuously from a peak production of 5 million tonnes in 1994.

Christensen *et al*. (2003) show that the biomass of top predator fishes in the North Atlantic has decreased by two-thirds in approximately 50 years. Similar declines were noted for other important species such as perch, anchovies, and flatfish as a result of overfishing between 1900 to 1999. However, the impact of overfishing goes beyond the impact on the populations of targeted species. One effect of overfishing is the progressive decrease of the trophic level of the catch. Overexploitation of the top of the food chain, leading to the targeting of more abundant species lower in the food chain, is called "fishing down the food chain" (Pauly and Watson, 2003). Overfishing has shortened the food chain and sometimes removed one or more of the links. This has increased the system's vulnerability to natural and human-induced stresses, as well as reducing the supply of fish for human consumption. In many cases restrictions on taking of smaller fish of each species has resulted in rapid evolution so that fish mature and reproduce at smaller sizes.

Livestock play an important role in the overall pressure of demand for fish. It is estimated than in 2004, 24.2 percent of world fishery production was used for fishmeal and fish oil for feed (Vannuccini, 2004). Approximately 17 percent of the fishmeal produced in the world is manufactured from trimmings from food fish processing and so has little independent impact on fish stocks. However, the remaining 83 percent comes from direct marine capture fisheries (Fishmeal Information Network, 2004). Fishmeal's importance as a feed component started in the 1950s in the United States industrial poultry production. It is now used as a feed ingredient in modern poultry and pig production, in developed and developing countries alike.

Fishmeal production increased until the mid-1980s and has been relatively constant at 67 million tonnes since then. As it takes 45 kilograms of wet fish to produce 1 kilogram of fish oil and dry fishmeal, this requires an annual ocean catch of 20–25 million tonnes of feed-grade fish, plus 4 million tonnes of trimmings from food fish (IFFO, 2006). To date, more than 80 percent of world fishmeal production originates in ten countries, of which the two largest producers are Peru (31 percent of the total) and Chile (15 percent). China, Thailand and the United States rank respectively third, fourth and fifth for production. At the same time, three Scandinavian countries (Denmark, Iceland and Norway), Japan and Spain rank respectively sixth to tenth. With more than 1 million tonnes per year, China is the largest world importer of fishmeal, followed by Germany, Japan and Taiwan (FAO, 2006b).

Currently, around 53 percent of global fishmeal production is used by the livestock sector (Fishmeal Information Network, 2004), 29 percent for pig production and 24 percent for poultry. Aquaculture is also a heavy user, and has expanded rapidly; it is now the fastest growing food producing industry in the world. Markets have reallocated the use of a fishmeal whose supply is limited. Between 1988 and 2000 the share of fishmeal consumed by the aquaculture sector more than trebled (from 10 percent to 35

© NOAA/C. ORTIZ ROJAS

About 400 tonnes of jack mackerel (Trachurus murphyi) are caught by a Chilean purse seiner – Peru 1997 percent), while the poultry sector's share more than halved (from 60 percent to 24 percent) (Tveteras and Tveteras, 2004). The reduced reliance on fishmeal in the poultry sector came as a result of nutrition research.

The shift towards aquaculture is presented by the fishmeal industry as "environmentally friendly" since fish are more efficient feed converters than terrestrial livestock (Shepherd *et al.*, 2005; Tidwell and Allan, 2001). But while the demand from the aquaculture sector will surely continue to rise (despite the fact that research effort is placed on reducing the share of this protein source in fish feed), there is little prospect for a further decrease in demand by the poultry sector. The strongly industrialized sector remains the fastest expanding livestock production segment, and already uses up-to-date nutrition know-how. In the meantime, demand for fishmeal from the pig production sector continues to increase (from 20 percent of global fishmeal supply in 1988 to 29 percent in 2000) (Tveteras and Tveteras, 2004). Fishmeal constitutes only a few percent of concentrate feed for monogastrics and this is unlikely to decrease further as it constitutes a highly valued protein input in the feed of these animals, particularly during the early stages (e.g. early weaned pigs).

The fishmeal industry claims that the recent stability of official fishmeal production figures is a result of fishery controls governing production, especially quotas, and that therefore there will be no increase in the future (Shepherd *et al.*, 2005). In view of the expected rise in demand, the enforcement of such regulations will need to be very strong. It may not be a coincidence that illegal, unregulated and unreported fishing has increased in many areas (UNEP, 2003). Fishing fleets are venturing farther from their home ports, off the continental shelves and into deeper waters to meet the global demand for fish (Pauly and Watson, 2003).

In the period 1990–1997, fish consumption increased by 31 percent while the supply from marine capture fisheries increased by only 9 percent (FAO, 1999). Some people suggest that this has intensified pressure on fishermen, which has translated into increased pressure on, and overfishing of, many commercial fisheries. Others say that pressure has been too high for a much longer period and that despite an increase in the reach and intensity of commercial fishing operations, the total quantity of fish catches is estimated (contrary to some official data – see GEO Indicators section, UNEP, 2003) to have been declining by about 700 000 tonnes a year since the late 1980s (Watson and Pauly, 2001). The initiatives to manage catches for specific fisheries have been ineffective in halting this downward trend. Alder and Lugten (2002) demonstrate for the North Atlantic that there has been a decline in landings, despite a plethora of agreements that focus on the management of stocks.

Whether global catches and global livestock fishmeal consumption increase or decrease, the latter clearly represents a substantial part of the former and hence the livestock sectors also bears considerable responsibility for the overexploitation of marine resources and the effect on marine biodiversity.

Erosion of livestock genetic diversity

The genetic resources embodied in domesticated animals have been strengthened by the breeding and selection efforts of farmers over thousands of years, in environments ranging from frozen tundra to hot semi-desert. Several thousand domestic animal breed[6] populations have been developed in the 12 000 years or so since the first livestock were domesticated, each adapted

[6] Breed is often accepted as a cultural rather than a biological or technical term. Genetic diversity measured at the molecular level does not always correspond to phenotypic breed diversity, because a long history of exchange, upgrading and crossbreeding has sometimes created similar genotypes with different phenotypes, or different genotypes within similar phenotypes. About half of genetic variability may be found between breeds but the share of within- and between-breed diversity varies among species and traits.

to specific environmental and farming conditions and each representing unique combinations of genes (Hoffmann and Scherf, 2006). Altogether more than 6 300 breeds of domesticated livestock have been identified.

This livestock genetic diversity is threatened. In 2000, over 1 300 of the breeds are now extinct or considered to be in danger of extinction. Many others have not been formally identified and may disappear before they are described. Europe records the highest percentage of breeds that are extinct or at risk (55 percent for mammalian and 69 percent for avian livestock breeds). Asia and Africa record only 14 percent and 18 percent respectively - however the data for developing countries in the World Watch List for Domestic Animal Diversity (Scherf, 2000) are much less complete than those for developed countries. Out of the 7 616 breeds recorded in the Global Databank for Farm Animal Genetic Resources, 20 percent is classified at risk (FAO, 2006b). When breeds without recorded population data are included, the number at risk may be as high as 2 255. These figures represent a 13 percent increase since 1993 (FAO, 2000).

This erosion of biodiversity is the result of what can be seen as competition among breeds, as the large number of specialized traditional breeds adapted to specific environments and cultures lose out to a greatly reduced number of modern commercial breeds. During the twentieth century, research and development in the commercial livestock sector has concentrated on a very small number of exotic breeds, with which rapid increases in meat, milk or egg production were achieved. This has been possible, because the environment in which these breeds perform has been drastically transformed and globally homogenized, removing or controlling the adverse climate, nutritional and disease effects that vary so much from one area to another. Only 14 of the approximately 30 domesticated mammalian and bird species now provide 90 percent of human food supply from animals (Hoffmann and Scherf, 2006).

This reduction in dominant breeds has gone to extraordinary lengths. Examples of specialized stocks are Leghorn chickens, which are superior for egg production, and Holstein-Friesian cattle, which dominate other dairy cattle breeds because of higher milk production (National Research Council, 1993). Over 90 percent of America's milk supply comes from Holstein-Friesian cows, while nine out of ten eggs come from White Leghorn hens. This focus is dictated by economies of scale, allowing for increased productivity gains by increasing the homogeneity of production and products through mass production.

Meanwhile, the genetic base of specialized traditional and regional stocks is narrowing because of a reduction in the effective population sizes as increasing numbers of producers shift to commercial breeds and the size of operations increases.

The arguments in favour of management and conservation of livestock genetic resources are the same as for other types of biodiversity: to maintain use and non-use values to humans,[7] to preserve important components of cultural heritage or typical landscapes, or to preserve traits that may be of value in the future. From the production point of view, the genetic pool is a source of material to confer disease resistance, productivity, or other properties sought after by consumer preferences (length and quality of wool, for example). The gene pool is also the basis for intensification; using conventional breeding techniques (other than genetic modification) it is quicker and more economic, to develop livestock by importing genes from outside a breed than by selecting within a breed. So breed diversity

[7] Use values indicate the direct value derived from food or fibre or other products or services, as well as the indirect value of contributing to landscapes or ecosystems. Another use value is the option value, which is the flexibility to cope with unexpected future events (e.g. climate or ecosystem change) or demands (e.g. disease resistance or product quality). Non-use value (existence value) is the satisfaction of individuals or societies stemming from the existence of the diversity.

allows more rapid genetic progress. Given that unpredictable challenges may emerge in future, from climate change to emerging diseases, a diverse gene pool will be essential for adapting to any change that may occur.

From the environmental viewpoint, however, conservation and further development of diversity may not always be exclusively beneficial. The pool of genetic resources potentially allows livestock to adapt to more demanding, currently too marginal, production environments, enabling them to adapt to a greater variety of habitats and increasing their environmental damage. It remains to be seen if livestock genetic, in balance, contributes to environmental resilience or degradation. Much depends on the management of the genetic resources.

5.3.5 Pollution

Over the past four decades, pollution has emerged as one of the most important drivers of ecosystem change in terrestrial, freshwater and coastal ecosystems. Like climate change, its impact is increasing very rapidly, leading to declining biodiversity across biomes (MEA, 2005b). Overall, pollution affects some 12 percent of globally threatened bird species (187 species), 29 percent of threatened amphibian species (529 species) and 4 percent (28 species) of the 760 threatened mammals for which data are available. The much higher percentage of threatened amphibians impacted by pollution than birds or mammals is probably a reflection of the larger number of species that are dependent on aquatic ecosystems where pollution is more pervasive. Pollution directly affects species through mortality, as well as through sublethal effects such as reduced fertility. Pollution can also have strong indirect effects by degrading habitats or reducing food supplies for animals.

The flow of nutrients (particularly nitrogen and phosphorous) from land-based activities into waterways and oceans is increasing globally. The predominant anthropogenic sources of nutrients are agricultural and industrial activities (fertilizer residues, wastes from animal husbandry, sewage, industrial effluents and atmospheric emissions).

The excess nutrient loads have led to eutrophication of lakes, rivers and coastal waters. Eutrophication involves the increased growth of phytoplankton and can favour the growth of toxic, or otherwise harmful, species. The decay of excessive plankton biomass increases the consumption of dissolved oxygen and occasionally causes periodic or permanent oxygen depletion, leading to mass mortality of fish and other organisms.

Pollution is potentially among the most damaging of all human influences on the oceans, in terms of both scale and consequences. Excessive nutrient inputs can turn marine areas into "dead zones" almost devoid of higher animal life. Nutrients discharged in large quantities into coastal waters promote blooms of planktonic and benthic algae. Phytoplankton blooms contribute to increased water turbidity, reducing light penetration and adversely affecting pelagic and benthic biological communities (GESAMP, 2001). Algal blooms involving toxin-producing species can cause the accumulation of algal toxins in shellfish to levels that can be lethal to other marine species and humans. The organisms affected by algal toxins are shellfish and finfish as well as other wildlife such as seabirds, sea otters, sea turtles, sea lions, manatees, dolphins and whales (Anderson *et al.*, 1993). Other adverse affects on ecosystem functioning were presented in Section 4.3.1.

Coral reefs and seagrass beds are particularly vulnerable to damage from eutrophication and nutrient loading. Eutrophication can also change the dynamics of these marine ecosystems and cause loss of biodiversity, including changes in the ecological structure of both planktonic and benthic communities, some of which may be harmful to fisheries (National Research Council, 2000).

Acid rain has been shown to decrease species diversity in lakes and streams. It has not yet been shown to be a significant issue in tropical freshwaters, where a large proportion of global freshwater diversity is found (World Conservation Monitoring Centre, 1998) - perhaps because industry is currently less developed in the tropics. However, depending on where the precipitation occurs, acidification of freshwaters can affect biodiversity at the species and subspecies level. The effects on freshwater fauna can be catastrophic. In Sweden alone, more than 6 000 lakes have been limed to preserve fish populations (Harvey, 2001).

As with the impact of climate change, the contribution of the livestock sector to global biodiversity loss from pollution is estimated to be proportional to its contribution to water pollution as presented in Chapter 4, which demonstrated that livestock have a major role in the pollution process through erosion and loading with pesticides, antibiotics, heavy metals and biological contaminants. The effect of soil pollution on soil biodiversity is not included in the following discussion because there is insufficient knowledge concerning the extent of soil pollution, soil biodiversity and the loss of soil biodiversity. It is safe to assume, however, that livestock-induced soil pollution is substantial in many locations, and soil is one of the most diverse habitats on earth. It contains some of the most diverse assemblages of living organisms. Nowhere in nature are species so densely packed as in soil communities: a single gram of soil may contain millions of individuals and several thousand species of bacteria.[8]

Direct toxicity from livestock–related residues and wastes

Pollution can act directly on organisms - basically by poisoning them - or indirectly by damaging their habitats. Pollution from livestock-related activities is no exception.

According to IUCN, perhaps the most dramatic recent example of the potentially devastating effects of direct toxicity of livestock-related pollution on wild species relates to vultures. In South Asia, vultures in the genus Gyps have declined by more than 95 percent in recent years owing to the toxic effects of the veterinary drug, Diclofenac, which is consumed when the birds feed on carcasses of livestock treated with the drug. Diclofenac is widely used in human medicine globally, but was introduced to the veterinary market on the Indian subcontinent during the early 1990s (Baillie, Hilton-Taylor and Stuart, 2004).

Residues of drugs used in livestock production, including antibiotics and hormones, have also been identified in various aquatic environments (Section 4.3.1). Low concentrations of antimicrobials exert a selective pressure in freshwater, allowing bacteria to develop resistance to antibiotics. Because this confers an evolutionary advantage, the related genes spread readily in bacterial ecosystems.

In the case of hormones, the environmental concern relates to their potential effects on crops and possible endocrine disruption in humans and wildlife (Miller, 2002). Use of hormones, for example, the steroid trenbolone acetate can remain in manure piles for more than 270 days, suggesting that water can be contaminated by hormonally active agents through runoff. The links between the use of hormones on livestock and their associated environmental impact is not easily demonstrated. Nevertheless, it would explain wildlife showing developmental, neurologic, and endocrine alterations, even after the ban of known estrogenic pesticides. This supposition is supported by the increasing number of reported cases of gender shifts in fish and the increased incidences of mammalian breast and testicular cancers and alterations of male genital tracts (Soto *et al.*, 2004).

Other livestock-related pollutants presented in Section 4.3 directly affect biodiversity as well. Water-borne bacterial and viral pathogens that

[8] See the FAO Soil Biodiversity Portal at http://www.fao.org/ag/AGL/agll/soilbiod/fao.stm for references.

affect wildlife species, and even livestock parasitic diseases, are transmitted, via water, to wildlife species. Chemicals such as chromium and sulphides from tanneries affect aquatic life locally, while pesticides have ecotoxicological effects for aquatic flora and fauna on a much larger scale. Although many pesticides dissipate rapidly through mineralization, some are very resistant and impact the health of wild animals and plants, causing cancers, tumours and lesions, disrupting immune and endocrine systems, modifying reproductive behaviours and producing teratogenic effects (i.e. causing malformations of an embryo or fetus).[9] With regard to pesticide use, Relyea (2004) tested the impact of four globally common pesticides on the biodiversity of aquatic communities: numerous species were eliminated and the ecological balance severely disrupted.

Pollution of habitats by livestock-related activities

Manure and mineral fertilizers used in feed production cause nutrient overloads in soils, as well as point and non-point source pollution of freshwater. Indirect eutrophication through volatilized ammonia is also important. Beyond consequences on local freshwater and soil habitats, the effects may reach as far as coral reefs. Emissions of sulphur and nitrogen oxides (SO_2, NO_x) from industrial livestock operations may also contribute to acid rain.

It is difficult to assess the effects of these forms of pollution on biodiversity. First, point-source pollution will be affected by the location of industrial livestock operations. Most industrial livestock operations (pigs, poultry and milk) are currently situated in peri-urban areas or locations with good feed supply, where biodiversity is generally low compared with wild areas. Second, as regards non-point sources, discharges and runoffs from pastures and livestock production units into main streams are mixed with other non-point sources. Therefore, their effects on biodiversity cannot often be dissociated from other forms of pollution and sediments.

Eutrophication of surface water damages wetlands and fragile coastal ecosystems, and fuels algae "blooms" that use up oxygen in the water, killing fish and other aquatic life (see Section 4.3.1 for other adverse effects). The contribution of the livestock sector to the rapidly increasing impact of eutrophication on biodiversity (MEA, 2005b) varies greatly around the world, but the importance of fertilizer use for feed production (Section 3.2.1) and the local importance of industrial livestock production units (Section 2.4) may well constitute good indicators for the regional importance of the sector's contribution. Based on the case of the United States analysed in Section 4.3.3, it may for example well be that the livestock sector as a driver of feed production has prime responsibility for the worsening of hypoxia (very low oxygen levels) in the northern Gulf of Mexico (see Box 5.6).

Threatened coastal habitats of East and Southeast Asia

Nowhere have the rapid growth of livestock production, and its impact on the environment, been more evident than in East and Southeast Asia. Over the decade of the 1990s alone, production of pigs and poultry almost doubled in China, Thailand and Viet Nam. By the year 2001, these three countries alone accounted for more than half the pigs and one-third of the chickens in the entire world. Not surprisingly, these same countries have also experienced rapid increases in pollution associated with concentrations of intensive livestock production. Pig and poultry operations concentrated in coastal areas of China, Viet Nam and Thailand are emerging as a major source of nutrient pollution of the South China Sea (FAO, 2004e). Along much of the densely populated coast, the pig density exceeds 100 animals per km^2 and agricultural lands are overloaded with huge nutrient surpluses.

[9] See also Chapter 4.

Box 5.6 Gulf of Mexico hypoxia[1]

The Mississippi River and northern Gulf of Mexico system is a prime example of the worldwide trend of increasing river-borne nutrients and the resulting diminution in the quality of coastal water.

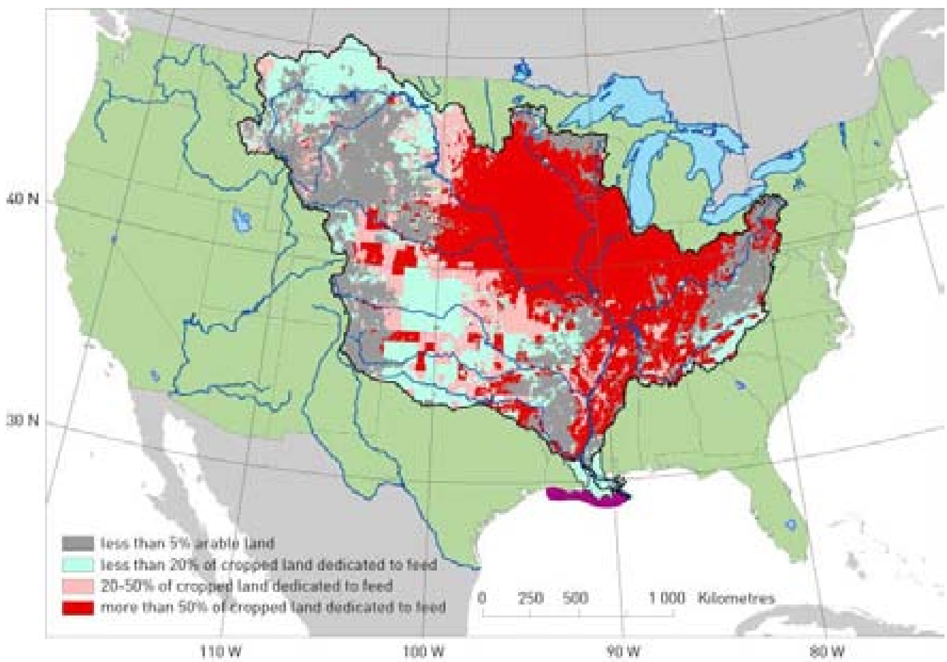
The Mississippi River system drains 41 percent of the contiguous United States into the Gulf of Mexico. It ranks among the world's top ten in length, freshwater discharge and sediment delivery (see Map 5.2).

The summer bottom-water hypoxic zone in the Gulf of Mexico has gradually grown to its present size, second in area only to the hypoxic zone of the Baltic basins (approximately 70 000 km^2). In midsummer 2001, the bottom-water area of the Gulf covered by hypoxia reached 20 700 km^2 (Rabalais, Turner and Scavia, 2002). Over this area, the level of oxygen fell to less than 2 mg/litre a level at which shrimp and demersal fish are not found. Hypoxia occurs usually only at the bottom near the sediments but can reach well up into the water column. Depending on the depth of the water and the location of the pycnocline (zone of rapid vertical density change), hypoxia typically affects 20 to 50 percent of the water column.

According to Rabalais *et al.* (2002) hypoxia might have existed at some level before the 1940–1950

[1] Hypoxia: a reduced concentration of dissolved oxygen in a water body leading to stress and death in aquatic organisms.

Map 5.2 Feed production in the Mississippi River drainage basin and general location of the 1999 midsummer hypoxic zone



Note: see Annex 3.4.
Source: adapted from Rabalais, Turner and Scavia, (2002).

Box 5.6 **cont.**

period; clearly it has intensified since then. For example *Quinqueloculina sp.* (a hypoxia-intolerant foraminiferan) was a conspicuous member of the fauna from 1700 to 1900, indicating that oxygen stress was not a problem at that time. Sediment core analyses also document increased eutrophication and organic matter sedimentation in bottom waters since the 1950s.

When polluted waters reach the ocean, much of the nitrogen will have denitrified by this point in the nitrogen "cascade." However, Rabalais and colleagues present compelling evidence for the close coupling of the levels of river-borne nutrients (nitrogen) and those of ocean primary production, net production, vertical carbon flux and hypoxia.

The analysis in Section 4.3.3 suggested that the livestock sector is the leading contributor to water pollution by nitrogen in the United States. In addition the Mississippi drainage basin contains almost all the United States feed production and industrial livestock production.

In light of these facts, the livestock sector may well bear the prime responsibility for worsening hypoxia in the northern Gulf of Mexico. This is confirmed by Donner (2006) who shows that a dietary shift away from grain-fed beef to vegetarianism in the United States could reduce total land and fertilizer demands of Mississippi Basin crops by over 50 percent, with no change in total production of human food protein. The change would return nitrate-nitrogen export by the Mississippi River to levels at which the Gulf of Mexico "dead zone" was small or non-existent.

Source: Rabalais *et al*. (2002).

Land-based nutrient pollution has caused algae blooms in the South China Sea, including one in 1998 that killed more than 80 percent of the fish in 100 km^2 along the coast of Hong Kong and southern China. These changes affect the habitats of many life forms, since the South China Sea supports substantial populations of fish, invertebrates, marine mammals and seabirds. The consequences for regional biodiversity may be far-reaching. As an example, since 2002 increasing masses of giant jellyfish reach the Japanese coast year round and severely hamper fishing campaigns. These species originate in the East China Sea, where they are proliferating because of an increasing availability of zooplankton resulting from land-based pollution induced eutrophication and decreasing fish stocks.

The impact of the decline in the quality of coastal seawater and sediment, in one of the world's most biologically diverse shallow water marine areas, the East Asian Seas, goes well beyond algal blooms and the related effects upon the food chain. Fragile coastal marine habitats are threatened, including coral reefs and sea grasses, which are irreplaceable reservoirs of biodiversity; the last refuge of many endangered species. Threatened coastal areas of the South China Sea, for example, have provided the habitat for 45 of the world's 51 mangrove species, almost all of the known coral species and 20 of 50 known sea grasses. In addition, the area is the world's centre for diversity of hermatypic corals, with more than 80 recorded genera, of which four appear to be endemic to the region; there are record high numbers of molluscs and shrimp species. It also contains a high diversity of lobsters, with the second highest endemism count (World Conservation Monitoring Centre, 1998). Southeast Asia contains one-quarter of the world's mapped reefs of which over 80 percent are at risk, and over half (56 percent) are at high risk. The most significant threats are overfishing, destructive fishing practices, sedimentation, and pollution associated with coastal development (Bryant *et al.*, 1998). Land-based pollution (industrialization, urbanization, sewage and agriculture) constitutes an increasing pressure on the coral reef ecosystems.

Pollution also drives habitat change in freshwater systems. Though eutrophication dramatically impacts locally, sediments from soil erosion, a non-point source pollutant caused by the livestock sector as well as by agriculture at large, are considered a larger threat. Section 4.3.3 discussed the numerous ways through which soil erosion impacts offsite habitats. Increased rates of sediment input into estuarine and coastal habitats have been observed (East Bay Municipal Utility District, 2001). Field studies have looked at the consequences of terrestrial sediment deposition, water-borne sediment and long-term changes in habitats. They indicate that (similar to the impact in freshwater ecosystems) increasing rates of sediment loading adversely affect the biodiversity and ecological value of estuarine and coastal ecosystems.

5.4 Summary of livestock impacts on biodiversity

We have attempted to present the full range of the more important and widespread impacts of livestock on biodiversity. Clearly livestock's shadow is very long: not only does it erode biodiversity through a wide range of distinct processes, but also its contribution to each of these processes takes multiple forms (e.g. Section 5.3.3). The shadow appears even larger if we consider that important ecosystem losses date back several centuries, with impacts still occurring today.

It is currently difficult to be precise when quantifying livestock-induced biodiversity loss. Losses are the result of a complex web of changes, occurring at different levels, each of which is affected by multiple agents. This complexity is further compounded by the consideration of the time dimension. In Europe, for example, practices such as extensive grazing that were responsible for much of the continent's historic habitat fragmentation are now seen as a means for conservation of today's much valued landscape (and sward) heterogeneity. Similarly in Africa, although pastoralists are responsible for past loss of wildlife through persecution of predators, pastoralism is often seen as a means to conserve the much needed mobility of remaining wildlife.

Nevertheless, we have attempted in this chapter to give an idea of the share of responsibility that livestock may carry for various types of loss and threat. Usually, this is based on our calculations in earlier chapters, for example on shares of greenhouse gas emissions, soil erosion or water pollution loads.

The processes can also be ranked in a more qualitative manner, according to their relative extent and severity. Table 5.3 presents such a ranking based on LEAD expert knowledge and the broad review of research results presented in this report. The large differences in impact between the losses related to extensive grazing and those to intensive livestock are reflected. The overall cumulative loss from extensive systems to date is much higher than that induced by the more intensive systems. This legacy is partly explained by the incomparably higher land requirements of extensive systems, and partly by the fact that intensive systems appeared only a few decades ago. The differences between the future trends (arrows in Table 5.3) show that for a number of processes, losses induced by intensive systems are increasing rapidly and may well surpass those that are more extensive. Some processes are related only to extensive systems (e.g. desertification), others to intensive systems (e.g. overfishing). In the past, the most dramatic losses were caused by extensive grazing, in the forms of forest fragmentation/deforestation and alien plant invasions, and by intensive systems in the form of habitat pollution.

Conversion of forest to pasture continuous to be an important process of biodiversity loss in Latin America, but this situation is rather atypical. At the global level, as described in Section

2.1.3, the land requirements of the livestock sector may soon reach a maximum and then decrease. More marginal land will revert back into (semi) natural habitat, and from there, under some circumstances, it may lead to the recovery of biodiversity.

Indications of the global impact of animal production and its distribution

International conservation organizations have collected vast amounts of data on the global status of biodiversity over the past decades. Data from organizations such as the WWF, the IUCN contain information on the nature of current threats to biodiversity (eg. Baillie, Hilton-Taylor and Stuart, 2004). These data collections, even though they do not cover the entire range of livestock related processes, provide clear evidence that the livestock sector's role in biodiversity erosion is very substantial.

An analysis for this report of the 825 terrestrial ecoregions identified by WWF shows that 306 of them reported livestock as one of the current threats – even though pollution from livestock is not considered, and important segments of the animal product food chain are ignored. The ecoregions threatened by livestock are found across all biomes and all eight biogeographical realms (see Map 29 in Annex 1).

The effect of livestock on biodiversity hotspots may indicate where livestock production is having the greatest impact on biodiversity. Conservation International has identified 35 global hotspots, which are characterized both by exceptional levels of plant endemism and by serious levels of habitat loss.[10] 23 of the 35 biodiversity hotspots are reported to be affected by livestock production (see Map 30 in Annex 1). The reported causes are related to habitat change and associated with the mechanisms of climate change, overexploitation and invasive alien species. Major reported threats are: conversion of natural land to pastures (including deforestation), planting of soybean for animal feed, introduction of exotic fodder plants, use of fire for pasture management, overgrazing, persecution of livestock predators and feral livestock. The role of the livestock sector in aquatic impacts (pollution and over-fishing) is not singled out.

An analysis for this report of the IUCN Red List of Threatened Species, the world's most authoritative source of information on extinction risk, indicates that the 10 percent of the world's species which face some degree of threat are suffering habitat loss from livestock production. Livestock production appears to have more impacts on terrestrial than on freshwater and marine species, as the important effects of habitat loss and habitat degradation are most significant on land.

5.5 Mitigation options for conservation of biodiversity

Classical approaches to conservation – such as attempting to preserve pristine habitats within national parks and other protected areas and to develop corridors between them – will always be necessary and will help to reduce the pressures on biodiversity. But in view of the severity and variety of current threats to biodiversity, efforts are also needed to reduce the many other pressures on wildlife. The livestock sector is a very significant source of many of these pressures, with a wide variety of impacts, many if not most of which occur in already disturbed environments.

Earlier chapters have described technical options for some of the specific threats which have an impact on biodiversity. In relation to wildlife, the focus should be on reducing those threats that currently have the largest impact or that are expected to become more important in the near future. Table 5.3 in the preceding

[10] The hotspot approach aims to identify the places where the most threatened biodiversity needs to receive the most urgent action. To qualify as a hotspot, a region must meet two strict criteria: it must contain at least 1 500 species of vascular plants (more than 0.5 percent of the world's total) as endemics and it must have lost at least 70 percent of its original habitat.

Table 5.3

Expert ranking of livestock- related threats to biodiversity resulting from the different mechanisms and types of production system

	Type of livestock production system		Affected level of biodiversity		
Mechanism of livestock sector induced biodiversity loss	**Extensive production**	**Intensive production**	**Intra-species**	**Inter-species**	**Eco-system**
Forest fragmentation	↗	↑	●	●	●
Land use intensification	↗	↑		●	
Desertification	→			●	
Forest transition (reversion of former pastures)	↗			●	●
Climate change	↗	↑	●	●	●
Invasive livestock	↘			●	
Plant invasions	↘	→		●	●
Competition with wildlife	↘	↑		●	
Overfishing		↗	●		
Livestock diversity erosion		↑	●		
Toxicity		↑	●		
Habitat pollution	→	↑		●	●

Legend: Relative level and type of threat to biodiversity resulting from the different mechanisms. "Extensive" and "Intensive" refer to the importance of the contributions from both sides of the continuum of livestock production systems.
Red shading indicates the level of past impact
■ very strong
■ strong
■ moderate
■ weak
white: no effect
Arrows indicate the direction of current trends
↘ decreasing
→ stable
↗ increasing
↑ rapidly increasing

section provides an idea of which processes and production systems may require most attention. Examples that stand out as important are the impact of land use intensification and habitat pollution induced by the intensive production environments; desertification in extensive grazing areas; and forest fragmentation related to both the extensive and intensive sectors.

In essence, mitigating the impact will consist partly in reducing the pressures, partly in better management of the interaction with natural resources, be it fisheries, wildlife, vegetation, land or water. The improvement of that management is more an issue of policy and regulation than of technical capacity building and research. Consolidating a network of well protected areas is an obvious start. This policy component of biodiversity conservation is dealt with in Chapter 6. Still for a number of threats technical options are available, which are presented here without discussion of the policy conditions required for their successful adoption.

To a large extent, biodiversity loss occurs as a consequence of environmental degradation processes analysed in the preceding chapters. Numerous options, highlighted in earlier chapter sections on mitigation, therefore also apply here for example on deforestation (also an issue of mitigating CO_2 emissions, Section 3.5.1), climate change (Section 3.5), desertification (rehabilitation of cultivated soils and pastures, Section 3.5.1; management of water, herds and grazing systems in 4.6), pollution (waste management and air pollution, Sections 3.5.3; 3.5.4 and 4.6.2).

A number of technical options could lessen the impact of intensive livestock production. Concerning feed cropping and intensive pasture management, integrated agriculture[11] provides a technology response by reducing pesticide and fertilizer losses. Conservation agriculture (see also Section 3.5.1) could restore important soil habitats and reduce degradation. Combining such local improvements with restoration or conservation of an ecological infrastructure at the landscape level (Sanderson *et al.*, 2003; Tabarelli and Gascon, 2005) and the adoption of good agricultural practices (sanitary measures, proper handling of seed lots avoiding contaminants, etc.) may offer a good way of reconciling the conservation of the functioning of ecosystems and the expansion of agricultural production.

Improvements in extensive livestock production systems can make a contribution to biodiversity conservation. Successfully tested options exist (see Sections 3.5.1 and 4.6.3) to restore some of the habitat lost by expansion of badly managed grazing land. In some contexts (e.g. Europe) extensive grazing may provide a tool to maintain a threatened but ecologically valuable level of landscape heterogeneity. Such options are commonly grouped under the denominator "silvopastoral systems" (including pasture management). Mosquera-Losada and colleagues (2004) present a wide range of such options and assess their effect on biodiversity.

These categories of options are all of great importance as they apply to wide-spread threats. Many others exist, often addressing threats of a more regional nature. Box 5.7 presents an example of a situation where the development of intensive farming of game species might contribute to the conservation of remaining wildlife.

It is important also to consider a more general principle. Land use intensification has been presented so far in this section as a threat to biodiversity because it is often synonymous with an uncontrolled profit-driven process with insufficient consideration for externalities (leading to loss of agro-ecosystem diversity). However, given the growth of the global livestock sector, intensification is also an important technological pathway, because it allows a reduction of the pressure on natural land and habitat, also reducing the risk of plant invasions.

[11] Integrated agriculture is a system of agricultural techniques developed in France in 1993 by Forum de l'Agriculture Raisonnée Respecteuse l'Environnement (FARRE). It is an attempt to reconcile agricultural methods with the principles of sustainable development, by balancing, in the words of FARRE, "food production, profitability, safety, animal welfare, social responsibility and environmental care."

Box 5.7 Livestock production to safeguard wildlife

Bushmeat was and remains an important inexpensive source of protein in African society. Hunting pressure on wild fauna has considerably increased over recent decades because of:

- population growth around forest and national parks has increased local demand for cheap and readily available meat;
- the development of the timber industry has opened many forested areas to settlers in areas where other sources of food supply may be less accessible. Settlers and timber industry workers may locally exert a significant level of hunting pressure on wildlife populations;
- hunting techniques improved massively during the twentieth century, with widespread diffusion of firearms and use of poisons; and
- the growth of urban centres creates an ever-increasing demand for meat supply as living standards improve.

The latter considerably modified the driving forces behind wildlife hunting and poaching. Urban demand evolves quickly, beginning with a demand for cheap protein to sustain food security, then adding on a demand for rare meats by the wealthy classes, who pay high prices. The bushmeat sector, though originally driven by subsistence needs from local actors, is increasingly driven by this economic rationale (Fargeot, 2004; Castel, 2004; Binot, Castel and Canon, 2006).

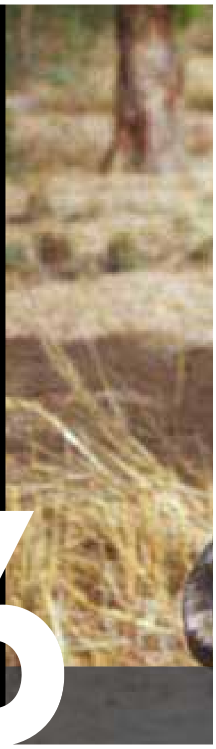
With the recent zoonosis crisis (Ebola, SARS), local consumers have changed their perception of bushmeat. Recent studies show that bush meat is no longer the food of preference for several local communities and temporary communities on the forest fringe (work forces hired by logging companies). Nevertheless, owing to the generally poor development of transport and marketing in the livestock sector in tropical Africa, availability of conventional meats is often too low – especially in areas where wildlife is at risk.

In this context, the livestock sector could help to lower the hunting pressure on wildlife by developing sufficient meat production and marketing capacity to guarantee food security and safety locally in areas where bushmeat consumption threatens wildlife. The development of an industrial livestock sector could supply the populations with meat at a cheaper price, but this is constrained by the lack of infrastructure. Carefully planned infrastructure development (transportation network, cold chain, etc.), to transport the products to the consumer or to transport production inputs (vaccines) required by livestock production units, might enable the livestock sector to contribute to wildlife conservation.

Non-traditional livestock production systems of selected wildlife species also offer alternatives to reduce hunting pressure on wildlife. The on-farm production of, the Greater Cane Rat (*Thryonomys swinderianus*) can be intensified, and can supply the urban centres with bushmeat. In rural areas "Game ranching" can provide regular bushmeat supply to the communities, regulating the market price of bushmeat and de facto reducing the poaching pressure on wildlife.

Sources: Houben, Edderai and Nzego (2004); Le Bel *et al.* (2004).

Adult greater cane rat (Thryonomys swinderianu) – Gabon 2003



06

Policy challenges and options

This chapter deals with the policy challenges and implications arising from livestock's growing and changing impact on the environment. First, the peculiarities of livestock-environment issues and the surrounding policy context are discussed and specific challenges identified. General policy requirements are identified for the livestock sector to address the basic environmental dimensions considered by this assessment: land degradation, climate change, water and biodiversity. Finally, specific policy options and practical applications will be presented that promise to alleviate some of livestock's environmental burden, viewed through the prism of the livestock-environment hotspots identified in Chapter 2.

The preceding chapters have established the body of evidence of livestock's large and growing impact on the environment. It has become clear that, for a large part, technical solutions already exist that could drastically reduce that impact. Why are so many of those solutions not widely applied?

Obstacles to effective livestock-environment policy making

It appears that two things are missing. First, there is a lack of understanding about the nature and extent of livestock's impact on the environment, among producers, consumers and policy-makers alike. Livestock-environment interactions are not easily understood. They are broad and complex, and many of the impacts are indirect and not obvious, so it is easy to underestimate livestock's impact on land and land use, climate change, water and biodiversity. Second – and partially as a result of the lack of understanding – a policy framework conducive to more environmentally benign practices simply does not exist in many cases, or is rudimentary at best. Often existing frameworks address multiple objectives and lack coherence. Worse still, existing policies often exacerbate livestock's impact on the environment.

Neglect may be sometimes conscious and deliberate. In many poor and middle income countries, food supply and food security, in their narrow definitions, are given priority over environmental concerns. There is solid evidence that relates environmental concern and the willingness to act for environmental protection to levels of income. The inverted U-shaped relationship between income and environmental degradation - rising at first as incomes rise, then as incomes rise further, starting to decline – has come to be known as the "environmental Kuznets curve" (see, for example, Dinda, 2005; or Andreoni and Chapman, 2001).

Neglect of environmental impact may sometimes be motivated by belief in the low chance of success of possible remedies. The hundreds of millions of poor livestock producers who, in the view of many, cannot possibly be expected to change their way of operating in the absence of alternative livelihoods, are probably the most striking example. The remoteness of livestock production in many of the world's marginal areas, and the difficulties in physically and institutionally accessing these areas, create practical problems even to establish the rule of law and the reach of regulation. Obvious examples of "lawlessness" in remote areas are squatters in the Amazon basin, or pastoralists in the "tribal" areas of Pakistan.

Neglect may also stem from the strong lobbying influence that livestock producers wield in many countries, particularly developed ones (Leonard, 2006). This affects the political economy of public policy making in the livestock sector in the EU, the USA, Argentina and elsewhere. It is often argued that in the past, livestock lobbies have been able to exert an over-proportional influence on public policies, to protect their interests. An indication of this lobbying power is the persistence of agricultural subsidies, amounting to an average of 32 percent of total farm income in OECD countries, with livestock products (dairy and beef, in particular) regularly figuring among the most heavily subsidized products.

Whatever the motivation, for the most part, livestock's impact on the environment does not receive an appropriate policy response even though the technical means to do so exist. At the low end of the intensity spectrum, in grazing areas in dry or otherwise marginal areas, in developing and developed countries alike, pastoralists and farmers are considered by policy-makers to be unable to afford to make or to maintain investments that could benefit the environment. At the high end of the spectrum, well-connected large-scale commercial producers often escape environmental regulations.

This neglect is in stark contrast with the magnitude of livestock's impact on the environment and underlines the importance and urgency of developing appropriate institutional and policy frameworks. Such frameworks should consist of economy-wide policies, sector policies for agriculture or livestock, and environmental policies.

6.1 Towards a conducive policy framework

6.1.1 General principles

A series of guiding principles need to be taken into account in designing and implementing policies to address livestock's impact on the environment. First we need to be aware of the principle sources of mistaken or misguided policy actions, including market failures, information failures and failures due to differences in political influence.

Rationale for government intervention

Public policies need to protect and enhance public goods, including the environment. The rationale for public policy intervention is based on the concept of market failures. These arise because many local and global ecosystems are public goods or "commons," and the negative environmental impacts that livestock have on them are "externalities" that arise because individual economic decisions usually consider only private individual costs and benefits. There are also consumption externalities through the negative health impact of excessive consumption of certain livestock products, particularly animal fats and red meat – however, these are beyond the scope of this study. Information failures also exist, for instance the inadequate understanding of highly complex phenomena such as biodiversity or climate change. As a consequence of externalities and information failures, the market fails to deliver a socially desirable level of environmental impact. Not only are there market and information failures, there are also policy failures, such as, for example, subsidies that sometimes constitute perverse incentives, promoting inefficient resource use or activities that damage the environment.

Market failures

With regard to livestock and the environment, most market failures occur in the form of externalities. These are impacts borne by third parties as a consequence of decisions by individuals or organizations, and for which no compensation is paid or received. Both negative and positive externalities exist. The presence of nitrates in water drawn from farmland, and the damage they cause or the cost of removing them from drinking water borne by a utility company, would be an example of a negative externality. The presence of wild birds in silvopastoral systems, the carbon sequestered on improved pasture, or reduced runoff and downstream sedimentation resulting from improved grazing management are examples of positive externality, through which a benefit is provided to society at large but for which usually no compensation is received.

Externalities give rise to economic inefficiencies, in that the perpetrator has little incentive to minimize the negative externalities, or to maximize the positive, because the consequences are borne (or enjoyed) by the society, not the individual or company responsible. Therefore, it is necessary for these external costs (or benefits) to be "internalized", that is, to create a feed-back mechanism for external impact to be accounted for by the perpetrator (or providers). The attempt to correct for externalities is represented by the "polluter pays, provider gets" principle.

The problem with applying this principle is that many environmental goods and services are not traded and, while they are obviously valued by society, they do not have a market price. In the absence of a market, valuing the environment in an appropriate way presents formidable challenges, (compare Hanley *et al.*, 2001; Tietenberg, 2003); and a host of methods have been developed. They are often distinguished into cost-based methods which try to assess the damage, the abatement costs or the costs of substitution of an environmental good or service; and demand-based methods which attempt to estimate the willingness to pay or other expressions of preference for environmental goods or services. Problems with valuation also become problems of policy design and implementation.

Policy Failures

Apart from market failures, another kind of inefficiency arises from the failure of government intervention, referred to as policy failure. As opposed to market failure, a policy failure represents a distortionary effect of active government intervention. Governments intervene in markets to achieve certain objectives. Policy failures may have adverse consequences, either by directly harming the environment or by distorting price signals and causing a misallocation of resources (FAO, 1999). Government interventions may fail to correct market failures, or they may make existing distortions worse, or sometimes create new distortions of their own. Policy failures can arise from sectoral subsidies, inappropriate pricing, taxation policies, price controls, regulations and other policy measures.

Next we need to consider some positive principles.

The precautionary principle

A principle frequently used to link environmental concerns to decision-making is the "precautionary principle", which calls for action to reduce environmental impact even before conclusive evidence of the exact nature and extent of such damage exists. The precautionary principle stresses that corrective action should not be postponed if there is a serious risk of irreversible damage, even though full scientific evidence may still be lacking. However, there is considerable debate about the usefulness of this principle among policy-makers, a common understanding is still missing (Immordino, 2003).

Policy level: subsidiarity principle

Environmental policies have local, national and global dimensions. Global issues such as climate change and loss of biodiversity have an international reach and are the subject of intergovernmental treaties. In view of the local nature of many livestock-environment interactions, the literature on environmental policy stresses the subsidiarity principle, i.e. that decisions should be taken at the lowest relevant organizational level and be as decentralized as possible.

The broader policy framework is usually set at the national level. Even international treaties on, for example, trade tariffs and emission targets usually need to go through a ratification process at the national level before becoming law. Regulations for emission control, taxation, agricultural and environmental subsidies are usually part of national policies. Local resource access management, zoning and enforcement usually fall upon local government authorities.

International decision-making – FAO, Italy

Policy process: inclusivity and participation

For policies to be successful, they need to be inclusive. At the local and national level, they need to involve, and possibly be designed by, all involved stakeholders. Their involvement enhances the chances that policies will be effective. The active participation of communities and citizens is required for local policies and projects, such as watershed protection, or the organization of farmer groups for technical assistance. However, in practice, participatory approaches seldom go beyond local activities. Usually participation does not extend into the design of sector-wide policy packages and development strategies (Norton, 2003).

Policy objectives and trade-offs: assessing costs and benefits

Livestock sector policies need to address a host of economic, social, environmental and health objectives. In most cases, it will be impossible to design policies that will address all at once and at reasonable costs to government and the people affected. Though important trade-offs exist and compromises need to be made. For example, land access restrictions and grazing controls on communal land often entail lower returns for grazers in the short run. Similarly, higher waste emission standards for intensive producers raise production costs and may affect the competitiveness of one country compared to others with no or lower standards.

Therefore, it is essential to carefully assess the costs and benefits of livestock sector policy interventions, and to prioritize different objectives. These will depend crucially on factors such as level of income and economic development, level of smallholder involvement in the livestock sector, prospects for livestock exports, extent of livestock-induced environmental degradation, level of market development and so on.

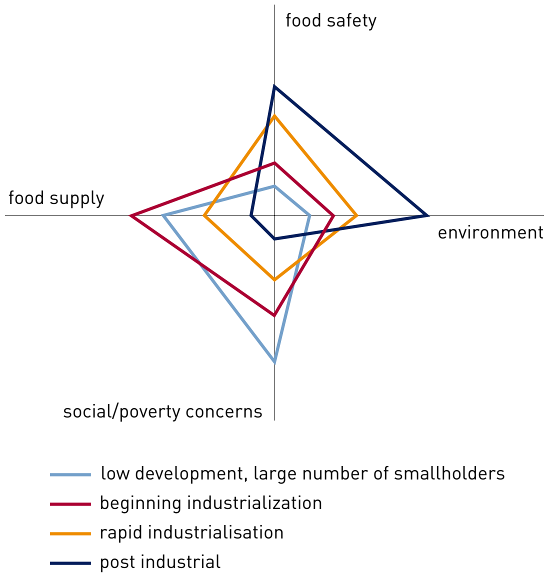
The four phases of development of policy priorities

Four different phases can be distinguished, depending on the level of economic development of a country.

Countries with low levels of income and economic development, and large involvement of smallholders in the livestock sector, often try to pursue social policies through the livestock sector, driven by concerns for the large masses of rural poor; other objectives are of second order. Most of sub-Saharan Africa and South Asia fall into this category. Typically, at this stage, policies include technology development and promotion, often in the area of animal production and health together with interventions in market development. The overriding objective is to maintain, and possibly further develop, the livestock sector as a source of income and employment for marginally productive rural people, as other sectors do not yet offer sufficient economic opportunities. Such strategies frequently fail to address, degradation and overexploitation of grazing resources, often under common property, in the form of overgrazing and other forms of unsustainable land management. Both governments and farmers lack the funds and ability to address widespread degradation. Regulatory frameworks may exist but are usually not enforced. Serious public and animal health issues relating to livestock are not vigorously addressed, either.

Moving up the ladder of economic development and income, into the early phases of industrialization, more attention tends to be given to environmental and public health objectives, but social objectives still maintain their predominance. Policy-makers are also concerned with the need to increase food supplies to growing cities. Allowing commercial meat, dairy and egg production in peri-urban areas provides a relatively quick fix. The smallholder livestock sector is still of overwhelming importance; although where livestock industrialization begins the smallholder sector tends to diminish in rela-

Figure 6.1 **Shift in livestock policy objectives in relation to economic development**



Source: Authors.

tive importance. The first attempts to address environmental objectives in the livestock sector are now being made. For example, by establishing institutions to deal with the degradation of common property resources, the establishment of protected areas, etc. Similarly, legal frameworks for food safety are being established and enforcement starts, usually with formal markets, and urban consumers begin to attract the attention of policy-makers. Currently Viet Nam may be a good example for this group and some wealthier African countries.

The picture changes more rapidly at the stage when developing countries fully industrialize. Governments no longer pursue social objectives in the livestock sector, as ample employment opportunities in secondary and tertiary sectors reduces the importance of the livestock sector as a social "reservoir", or "waiting room for development". On the contrary, a number of countries, such as Malaysia, actively encourage the demise of smallholder agriculture to mobilize additional labour for industrial development, and to rationalize the agro-food industry. Food safety standards are established to satisfy rapidly growing cities' increasingly sophisticated bulk demand for meat, milk and eggs. The ensuing consolidation of the food industry quickly reduces the number of producers and other market agents.

At this stage, the livestock industry becomes a profitable business and consolidates. The sector is increasingly expected to meet basic environmental standards, as the public begins to perceive the elevated environmental costs of rapid industrial development. However, agricultural and livestock lobbies sometimes maintain their influence and achieve protection, as a legacy of the sector's past importance, or because of the importance assigned to self-sufficiency in food products, or because of the cultural values embodied in livestock. Many East Asian countries such as China and Thailand, and Latin American countries such as Brazil and Mexico, are examples of this stage, even though these countries are highly diverse and heterogenous.

At full industrialization, environmental and public health objectives take predominance. The livestock sector is much reduced in its relative social and economic importance. However, in most OECD countries the agricultural and livestock sector is still more important in terms of employment than in it is in terms of contribution to GDP, and the agricultural sector regains some importance for services other than the provision of food and other primary products. The level of protection for livestock commodities indicates, for most developed countries, that related lobbies still wield widespread influence over policy-making.

Taking these observations into the future, it is not difficult to imagine the next step in fact, it is already taking shape. The demands for environmental services against the background of increased food supply, driven by heightened, and ever more sophisticated, consumer expectations will establish environmental and food safety requirements as the only motives in public policy-making. Protection will wane and implicit rights gradually disappear.

The stylized pattern of the four stages and their changing priorities is depicted in Figure 6.1. While no attempt is made to provide statistical evidence for these observations in the context of this study, such considerations are explicit in multi-criteria and hierarchical decision-support tools, such as in Gerber *et al*. (2005). The implicit trade-offs indicate that it may not be realistic to expect - as many in the livestock research and development community do - that the livestock sector can deliver on economic, social, health, and environmental objectives all at once and in a balanced form. Tools like hierarchical or multi-criteria decision-making can help addressing these trade-offs, but the conflicted and distorted policy framework, within which the livestock sector operates, is not easily disentangled.

The important subsidies that most developed countries have provided to the livestock sector underline the fact that the sector is assigned importance beyond its mere economic contribution. It can be stated, therefore, that the livestock sector continues to receive the attention of policy-makers for social, economic and food safety reasons, and the trade-offs that exist between these three and the environmental objectives often work to the detriment of the latter. The reasons for this vary, depending on the stage of development, but the overall tendency seems to be very widespread.

There may be a causal link between government subsidies and natural resource degradation. Chapters 3 to 5 give a description of what we might call "nature's subsidies" to the livestock sector - the provision of natural resources and waste sinks and their gradual degradation or exhaustion, without restoration or remediation. Eliminating a large part of these subsidies is a requirement for better resource use and limiting livestock's impact on the environment.

However, there will be a price to be paid:

- Consumer prices for livestock products are likely to go up as a result of correcting input prices for water and land, especially prices of beef and other types of red meat. Nature's subsidies are particularly high for ruminant products (in addition to high government subsidies in OECD countries).
- Livestock farming in many marginal areas, under common and private property alike, will often become unprofitable if current price distortions are removed and externalities are factored in. Many producers will need to find alternative livelihoods. If it is accepted that this is a desired long-term outcome, policies need to change direction now.
- The drive towards higher efficiencies, which will also generate savings in use of natural resources and reduce emissions, will make livestock production increasingly knowledge- and capital intensive. As a result, small family-based livestock producers will find it increasingly difficult to stay in the market, unless effective organizational arrangements, such as contract farming or cooperatives, can be designed and used (Delgado and Narrod, 2002). Again, the loss of competitiveness requires policy interventions, not necessarily to maintain smallholder involvement in agriculture, but to provide opportunities for finding livelihoods outside the agricultural sector and to enable an orderly transition.

Broad policy approaches: regulatory and economic instruments

Usually, policies do not consist of a single measure but of a series of measures. The key to successful policy design and implementation often lies in ensuring the right mix and sequencing of different policy measures.

Generally, the literature distinguishes between two broad approaches for implementing environmental policies: regulatory approaches and economic instruments. The choice between these approaches is not merely ideological, it also depends on the capacity of governments to enforce regulations; and wide differences exist between countries.

- Regulatory approaches (often termed "command and control") are often applied to emissions into the air, water and soil (mostly in

cases of point-source pollution) and generally, for access to and use of resources. Such approaches rely on sometimes onerous monitoring and enforcement, and depend on the related institutional capacity, which limits their use in many developing countries. Historically, environmental policies in most countries have started off with "command and control".
- Economic instruments rely on the role of monetary incentives to modify the behaviour of individuals or companies. They can be positive (in the form of subsidies or revenues from the sale of environmental services) or negative (in the form of levies or taxes). Many instruments rely on economic efficiency as the basic objective. Monitoring costs for economic instruments tend to be lower as there is greater scope for self-regulation, rewarded by financial incentives.

Commonly, both these approaches are used in combination. Other policy instruments include technology support and related capacity building, institutional development and infrastructure development.

Policies can drive changes in technology and management

Policies define rights and obligations. They also have the potential to determine input and output prices, and thus drive the delivery of public goods towards what society considers to be the optimal level. The concept of "induced innovation" widely published by Hayami and Ruttan has proved useful in the context of livestock–environment interactions (de Haan, Steinfeld and Blackburn, 1997). Ruttan (2001) links this concept to an earlier observation by Hicks (1932, pp. 123-25):

> *"A change in the relative prices of the factors of production is itself a spur to invention and to inventions of a particular kind – directed at economizing the use of a factor which has become relatively expensive."*

The induced innovation concept has since been further developed to include institutional change; for example Coase and Williamson (McCann, 2004) suggest that forms of economic organization, such as vertical integration, are the result of minimizing transaction costs. Without going into further detail of the economic models underlying these concepts, it is useful to view policies as potentially powerful drivers of technological change through their effect upon prices and their regulation of access to resources. By restricting access to grazing land, for example, land and related feed resources become relatively scarce, so technical change will move towards making more efficient use of these resources. Likewise, better pricing will encourage more efficient use of water, and drive water use towards optimal allocation among different competing uses (livestock, crops and other). The same applies to all other natural resources that feed into the livestock production process, such as water or nutrients. Likewise, new costs associated with the internalization of externalities from livestock production, such as emissions of ammonia or other forms of waste, will lead to increased efforts towards their avoidance. These effects are likely to be all the more important the higher current differences are between actual costs or prices and those reflecting an "optimal" level of environmental protection.

Today's decision-making on the livestock–environment–people nexus is characterized by the severe under-pricing of virtually all natural processes that go into the livestock production process, by the neglect of major down-stream externalities generated by the livestock sector without it being held accountable; and by a number of distortions, creating (broadly speaking) subsidized livestock sectors in developed countries and taxed ones in developing countries. Decision-making is further complicated by unrealistic expectations about pursuing social objectives through the livestock sector.

To summarise, the canvas upon which new policies will be designed is not blank, as it is already marked with broad brush strokes resulting from ignorance, neglect, conjectures and fallacies. This should not give rise to despair - rather it should inspire hope that relatively minor changes, in a sector that has often been considered environmentally unimportant, could have a major impact.

6.1.2 Specific policy instruments

Limiting livestock's land requirements

One important key to limiting livestock's environmental impact is to limit livestock's land requirements by pitching policies within the context of the geographic transition that the livestock sector is undergoing. As we have seen in Chapter 2, this transition has two facets.

First, there is the expansion of land used by or for, livestock. Until the mid-twentieth century, this was mainly in the form of grazing land. This expansion is still continuing in sub-Saharan Africa and especially in Latin America, where pasture is the main follower of deforestation. However, in most parts of the world, this expansion has either come to a halt (Asia, the Near East) or gone into reverse, with pasture reverting back to woodland or forest (industrialized countries).

At the same time, the use of concentrate feed has expanded significantly over the last 50 years greatly increasing livestock's demand for arable land. As of 2001, an estimated 33 percent of total arable land is devoted to producing feed, either as primary commodities (grains, oilcrops, tubers) or their by-products (brans, cakes). Again, this area expansion, although still ongoing in most developing countries, is poised to slow down and eventually reverse. This is happening already in industrialized countries where stagnant or modestly increasing demand for livestock products is accompanied by continuous gains in livestock productivity and crop productivity, resulting in lower overall land requirements for livestock.

If overall land requirements can be further reduced, which seems possible, this will benefit the environment by freeing land for environmental purposes. It would need to be accompanied by careful intensification of existing grazing


© FAO/21424/J. SPAULL

An example of urban animal husbandry showing goats grazing on the citadel in the centre of Amman. Jordan - 1999

and arable land, where the potential for yield increases exists.

Second, there is the growing concentration of livestock activities in certain favoured locations. This applies to the industrialized parts of the livestock sector, notably intensive poultry and pig production and, to a certain extent, dairy and beef. As we have seen, this concentration is driven by the newly gained independence of industrial livestock from the specific natural endowments of given locations, which have previously determined the location of livestock production (as they still do for most of crop agriculture).

Geographic concentration, or what could be called the "urbanization of livestock," is in many ways a response to the rapid urbanization of human populations. Peri-urban livestock provides a quick fix for countries in rapid economic development with fast-growing urban centres. This geographic concentration is largely responsible for the problems related to disposing of livestock wastes by recycling on surrounding land.

However, developed countries have been relocating their livestock production away from cities, and have established infrastructures and regulations to do so. The same is happening in emerging economies, first as a response to the nuisance factors of livestock (odour and flies) and then to the issues of nutrient loading of waterways and public health. Policies are needed in emerging economies to facilitate rural-based livestock industries, and to avoid the "urbanization of livestock" where it has not yet occurred.

In the following sections, basic policy instruments, currently applied and possible responses to livestock's role in environmental degradation, are described along with their requirements and potential impact. The choice of policy instruments needs to be based on their efficiency; that is the level pollution control resources are extracted at which the difference between social benefits and social costs is maximized (Hahn, Olmstead and Stavins, 2003). Increasingly, however, the efficiency criterion alone is being complemented by effectiveness considerations. These begin with an environmental objective (such as the level of nitrates in drinking-water) and then the attempt to achieve the target at minimum aggregate cost, often including market-based instruments so as to bring about an allocation of at least the cost of pollution reduction. Another criterion to be used in the choice of policy instruments is that of equity, since the distribution of pollution control costs and environmental benefits is often unequal (Hahn, Olmstead and Stavins, 2003).

Correcting distorted prices

Many of the inefficient, degrading, wasteful or otherwise damaging aspects of livestock production result from distorted price signals that discourage efficient resource use and foster misallocation and uncontrolled degradation of resources. This relates in particular to underpriced natural resources and sinks, either as a result of an overt subsidy (as for example in the case of water) or because of a disregard for externalities.

Largely, market failures and policy distortions mean that current prices for inputs and outputs of livestock production do not reflect true scarcities. As we have seen in Chapter 3, the livestock sector is highly dependent on natural resources such as land, water, energy and nutrients. Yet these resources are almost universally underpriced because of policy distortions or because externalities are unaccounted.

Land is the most important factor of agricultural and livestock production. Land taxes are seen as an instrument to encourage more productive or intensive use of land. Particularly, land taxes may counteract speculation in situations where owners hold land, not for productive purposes, but as an asset to hedge against inflation, which is common in some Latin American countries (Brazil, Costa Rica) (Margulis, 2004). Further, land taxes may induce more efficient utilization of land and encourage its redistribution, since smallholdings tend to be more land-intensive and achieve higher yields (Rao, 1989).

Strengthening land titles

Without clearly defined rights of access to land, incentives are weakened for livestock and crop production to be carried out in a way that maintain the land's long-term productivity. Land and land-tenure policies are usually considered in light of goals concerning economic efficiency and the objectives of equity and poverty alleviation; although environmental issues are of increasing importance. Given the increasing scarcity of suitable agricultural land in most parts of the world, and the growing concerns about deforestation and land degradation, increases in land productivity will have to continue to provide the bulk of increased food supplies.

While most of the area cropped for feed is under private ownership, a large part of ruminant livestock production still takes place on communal lands (such as most of sub-Saharan Africa) or state lands (such as in India, Western Australia and Western United States). There seems to be a wide consensus that land titling and secure access to land, such as the long-term land leases practiced in China, are a prerequisite for agricultural intensification, gradual transition to full titling is occurring in response to population pressure. Norton (2003) states that "in regions of the world where customary rights already had been weakened or superseded, and where the State is not the sole owner of agricultural land, the case for accelerated implementation of titling systems is strong." Land titling is seen as a prerequisite to private investment in land, including those that protect and enhance its long-term productivity and those that benefit the wider environment.

Pricing water realistically

With regard to water Pearce (2002) estimates that between 1994 and 1998 annual water subsidies in developing countries amounted to US$45 billion per year. Water in agriculture is severely under-priced. Water has been identified as a major resource for livestock production, whether in the form of "blue water" (for irrigating fodder or feedcrops, for drinking, for waste management or for product processing), or in the form of "green water" - water on rainfed pastures that translates into vegetative growth for livestock grazing. The latter's importance is further enhanced by the essential function of many grassland areas in harvesting water and regulating its movement - both of which are crucial in providing reliable freshwater supplies for growing urban, industrial or agricultural needs.

The push towards efficiency, equity and sustainability in agricultural water management needs to be put into a broader context. As Norton (2003) puts it "achieving greater efficiency in irrigation in the broader sense may mean giving up water to other sectors where it has higher-value uses, even if sometimes that implies reducing the value of agricultural output." What holds true for irrigation is certainly true for all agricultural uses. Except for where irrigation water is used for forage crops, as in some OECD countries, livestock's use of freshwater does not often create a high level of agricultural output per unit of water, particularly when most of this water is used to keep animals alive rather than for producing output.

The fact that water is so widely and severely under-priced entails that water use is less efficient than would otherwise be the case. If prices were higher, water would be allocated differently as between agricultural uses and other uses. In stark contrast to current practices, Bromley (2000) calls for water pricing to be seen as part of a regime in which farmers are induced to contribute to a public good for several important goals to:

- stimulate conservation of water;
- encourage allocation to its highest value use (including non-agricultural uses);
- minimize the environmental problems arising from inefficient irrigation;
- generate enough revenues to cover operating and maintenance costs; and
- to recover the original investment.

Various methods are used for pricing water (Tsur and Dinar, 1997) including: volumetric, output, input and area (see Section 6.1.4). Formal markets for water rights currently exist in only a few places (such as the Australia, Brazil, Mexico and the western United States). In recent years, general interest has been excited because of their potential to foster efficient use of an increasingly scarce resource (Norton, 2003). Water markets work on the basis of legally recognized and registered water use rights. These rights are separate from land titles, and individuals and groups can trade water rights within the scheme. While there are a series of conceptual and location-specific practical issues, water markets have the potential to provide incentives for conserving water and to allocate it to higher-value uses. Through water pricing governments can monitor operations, more easily enforce regulations and prevent the abuse of monopoly power (Thobani, 1996).

There are similar price distortions where livestock are used for other than productive use. As described in Chapter 2, livestock are used to acquire land titles, leading to or contributing to deforestation. Likewise, livestock are used as an asset or as a store of wealth in many grazing areas under common property regimes, leading to or contributing to overgrazing. Both are cases where non-productive uses of livestock have taken predominance, and the ensuing resource degradation is a reflection of market imperfections and institutional failures. Removing price distortions and pricing natural resources at their actual cost will generally increase production costs and may thereby reduce overall consumption levels for animal products and livestock related services.

Removing subsidies can reduce environmental damage

In the livestock sector of most developed and some developing countries, subsidies strongly distort prices at the input and product level. In all OECD countries, in 2004, subsidies to agricultural producers amounted to more than US$225 billion a year, equivalent to 31 percent of farm income. There is increasing evidence that subsidies are not neutral in terms of environmental impact and, indeed, that certain forms of subsidies generate negative environmental effects (Mayrand *et al*., 2003).

For some countries, the removal of subsidies has been shown to have a strong potential to correct some of the environmental damage caused by livestock production. For example, New Zealand (see Box 6.1) made sweeping subsidy reforms in the 1980s, and now reports that the removal of subsidies resulted in significant reductions of environmental damage caused by agriculture in general, in the form of increasing forest land, less erosion, and less nutrient runoff. In the livestock sector in particular, it led to reduced grazing pressure in the hill country of the Northern Island (MAF-NZ, 2005).

Mayrand *et al*. (2003) and UNEP (2001) have used the OECD methodology (developed for assessing the environmental impacts of trade liberalization OECD, 2001) to asses the environmental impacts of agricultural subsidies. The authors found that subsidies had a significant impact on the environment, through their impact on scales of production, the structure of agriculture, input and output mixes, the technology of production and the regulatory framework.

Particular forms of impact include:

- Market price supports affect the scale of production. They translate into higher and more

Box 6.1 **New Zealand – environmental impact of major agricultural policy reforms**

In 1984, the New Zealand Government changed the agricultural policy almost overnight from one of heavy protection and subsidy (for example, in 1984, the assistance payment to farmers for lamb was 67 percent of the farm-gate price) to one of the most open, market-oriented agricultural sectors in the world. Export subsidies were eliminated and import tariffs phased out. Output price assistance for agricultural products and subsequently, fertilizer and other input subsidies were abolished. In addition, tax concessions to farmers were withdrawn. Free government services for farmers were eliminated.

While the first years were particularly stressful for the rural sector, very few farmers were forced by the reforms to leave the land. The rural collapse predicted by some never happened. New Zealand's rural population rose slightly between the 1981 census and the 1991 census despite the removal of subsidies. Since the removal of agricultural subsidies in the mid-1980s, there has been a gradual but steady change of land use from pastoral agriculture to forestry. Total area in various forms of pasture has declined from 14.1 million hectares in 1983 to 13.5 million hectares in 1995 and to 12.3 million in 2004. Meanwhile, the area of planted forest has increased from 1.0 million to over 1.5 million hectares, a 50 percent increase, over the same period, and to 2.1 million in 2004. Fertilizer use declined in the first decade after the reforms, and, there is some evidence of reduction in leaching of phosphates from hill country pasture catchments, where phosphate is the dominant nutrient applied. Soil erosion has also declined leading to improved water quality. However, the increased use of nitrogen fertilizer, following the move to dairy production, is a more worrisome trend.

Sources: MAF websites and Harris and Rae (2006).

intensive production levels. This affects the environment through input use (water withdrawal, fertilizer applications, etc.) and area expansion (for crop agriculture) or expansion of livestock numbers. The OECD (2004, p. 19) found that "in general, the more a policy measure provides an incentive to increase production of specific agricultural commodities, the greater is the incentive towards monoculture, intensification, or bringing marginal (environmentally sensitive) land into production, and the higher is the pressure on the environment".

- Support to agriculture can distort the allocation of resources because it is often unequal across commodities. In the livestock sector this can be exemplified by the high support to dairy as opposed to the small subsidies for poultry. As a result, farmers concentrate on the production of the most subsidized commodities, leading to reduced cropping flexibility and increased specialization. This in turn tends to decrease agricultural and environmental diversity and to increase the vulnerability of agro-ecosystems. An example is provided by the imposition of milk quotas in many OECD countries for price stabilization, which led to a geographic concentration of milk production (OECD, 2004, p. 20). Together with higher milk prices, farmers attempted to maintain profit levels by cutting production costs, reducing the number of cows while increasing their yield. This resulted in higher input use (feed concentrates) and reduced grazing, thereby increasing the intensity of dairy production and aggravating environmental pressures in specific locations.
- Subsides can prevent technological change by supporting specific inputs or technologies - thereby creating a technology "lock-in" effect (Pieters, 2002). For example, in the EU high price supports for cereals drove livestock feeding towards the use of cheaper cassava in the 1980s and 1990s thereby preventing advances in cereal feeding that would otherwise have occurred, and causing a massive transfer of nutrients (de Haan, Steinfeld and Blackburn, 1997). On the other hand, removal of such subsidies could induce technological change with more positive environmental outcomes. Also, shifting from subsidies for production towards payments to farmers for environmental services can lead to enhanced environmental benefits.
- It is generally accepted that agricultural subsidies affect the structure of agriculture, the number and size of production units and the organization of the value chain (e.g. vertical integration). However, both subsidies and trade liberalization are said to work towards large-scale industrial agriculture.
- Subsidies also have a distributive impact. A recent study by the OECD (2006) found that a large share of farm subsidies end up supporting land owners and input suppliers. When they are based on production totals, they tend to benefit larger farms and impoverish smaller ones and drive them out of business.
- Trade reforms may have a regulatory effect, i.e. they may have an impact on environmental regulations and standards. This may work both ways: on the positive side, agreements on trade liberalization may include measures to improve environmental standards. On the negative side, particular provisions of trade reforms may limit a country's ability to observe environmental protection standards (UNEP, 2001).

Mayrand *et al*. (2003) also found that market price support (which accounts for two-thirds of total subsidies in the OECD) is among the type of subsidy most likely to generate perverse environmental impacts. Market price support is included in the "amber box" of the Doha round of trade negotiations (the amber box includes support that should be reduced or removed, including all domestic support measures "considered to distort production and trade"). There is increasing evidence that the reduction of amber box subsidies can constitute both a trade liberalization and a benefit for the environment. Also, other types of subsidies (payments based on inputs, for example) tend to have a more neutral and sometimes positive impact on the environment. The OECD (2004) came to the same conclusion, in a review of policies and their impact on agriculture and the environment. Despite some reforms, agricultural support linked to production remains the predominant form of support in OECD countries. The OECD work shows that this provides incentives to adopt environmentally harmful practices and to expand production into environmentally sensitive land. The OECD also deplores the lack of policy coherence, with agro-environmental measures and commodity production-linked support policies pulling in opposite directions.

Trade liberalization and its environmental impacts

Rae and Strutt (2003) came to a similar conclusion when attempting to assess environmental pollution from livestock as affected by trade liberalization in OECD countries. They used the OECD nitrogen balance database in conjunction with a global computable general equilibrium model. Using three different scenarios of increased trade liberalization, their computations all resulted in improved environmental outcomes, with a reduction in the surplus nitrogen that can cause damage to soil, air and water. Rae and Strutt found that "total OECD nitrogen balances are expected to fall more, the more ambitious the reform modelled" (Rae and Strutt, 2003; p.12). In contrast, Porter (2003) argued in the case of the maize/beef sector that the production effect (the expansion of a commodity sector in response to positive price signals) as a result of trade liberalization is rather limited. He found that the environmental impact, stemming from expansion, is mediated or even nullified by technology advance. In addition, reactions to price signals are severely conditioned by the long "cattle cycle", i.e. the time lag between herd management decisions and bringing cattle to the market. However, this observation may be limited to the beef sector.

While trade liberalization seems to offer opportunities for reducing the environmental impact of livestock, there are various trade-offs, and complementary measures may be needed. First, trade liberalization will result in increased trade and hence movement of goods, which has its own environmental costs. These can sometimes offset any gains resulting from better resource use at the production level. Second, trade liberalization will likely be accompanied by locational shifts of livestock production to less densely populated areas, hence to accompany the shift, environmental polices are needed in areas where livestock production is growing. For example, Saunders *et al.* (2004) investigated environmental impacts of dairy trade liberalization through the application of a multi-commodity, partial equilibrium model for OECD countries. Their results "support the notion that production and environmental heterogeneity both between and within trading partners will lead to spatially differential changes in pattern of resource usage and environmental impacts (Saunders, Cagatay and Moxey, 2004, p.15).

More generally, trade-related policies and other macro-economic policies such as devaluation, commodity price stabilization, preferential trading arrangements all tend to have a significant impact on the environment (UNEP, 2001, p. 17). Environmental policies can be seen as second order policies, which are brought in after the gross macro-economic and trade policy distortions have been corrected.

What are the alternatives to commodity production-linked support? Various policy measures are being applied and studied, mostly in OECD countries:

- In some countries, land set-aside schemes are being applied that provide farmers with an incentive to set aside their poorest, economically marginal land. Here, the environmental impact crucially depends on quality of the natural resources associated with the land set aside. The more valuable the land in environmental terms, and the lower its value in productive terms, the more successful these schemes have been.

- Increasingly, production-linked support measures are linked to a requirement to meet certain environmental targets, known as cross-compliance. A recent OECD publication (2004) states that cross-compliance allows for a better harmonization of agricultural and environmental policies. It also may increase public acceptance of support to agriculture. However, any change in the level of support will change the effectiveness of cross-compliance, which carries the risk of losing environmental leverage when production-linked support is reduced. Adherence with cross-compliance requirements is also difficult to measure.
- Part of "getting the prices right" is the need to compensate livestock farmers for the environmental benefits they provide. The most frequent example is managing grazing pressure in water catchment areas to improve water infiltration and reduce siltation of waterways. A LEAD-initiated project in Central America experiments with payment for environmental services generated by improved pasture and silvo-pastoral systems, particularly improved biodiversity and carbon sequestration (see Box 6.2).
- In the case of environmental issues related to pesticide use, water quality, ammonia and greenhouse gas emissions, agro-environmental measures continue to focus on setting standards and targets.
- Pollution issues, such as manure storage and application, are subject to regulations governing related practices (mode and time of application, for example), and are supported by fines and charges for non-compliance.

Compared with other sectors, the agricultural sector is characterized by a relative absence of environmental taxes and charges and the dominance of incentive payments. This suggests that farmers have strong political clout and have succeeded in creating political acceptance of their implicit or "presumptive" rights in the use of natural resources. Therefore, there is still a wide scope for better cost internalization to correct for environmental damage and encourage pollution treatment.

Regulations

Regulations typically specify technologies or uniform emission limits. Regulations are the policy instrument of choice at the early stages of addressing environmental objectives. However, their implementation requires institutions for monitoring and enforcement. This is particularly difficult in remote and poor areas, and when dealing with non-point source pollution. In contrast, where pollution is highly localized and where livestock production is commercial, the prospect of enforcing regulations is improved.

In extensive livestock production, regulations are frequently established to limit grazing pressure or to protect environmentally sensitive areas. While grazing restrictions operate successfully in many cases in developed countries, success has been rather limited in developing countries unless there are strong local organizations.

Regulations concerning water are often used to set emission standards for the control of pollution from livestock activities. These are discussed in more detail in Section 6.1.3. Environmental regulations affect the spatial distribution of livestock; for example in the United States, Isik (2004) shows that areas with more stringent environmental regulations suffered declines in livestock numbers to counties and states with less stringent regulation (called "pollution havens").

A number of countries have started to address air-pollution related to the issues of nitrous oxide emissions and ammonia volatilization by means of regulations.

At the international level, the United Nations Economic Commission for Europe's Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (also known as the Gothenburg Protocol) was signed in 1999, under the 1979 Geneva Convention on Long-range Transboundary Air Pollution. It entered into force in May 2005. The main signatories are the European Community, the individual European countries, the USA, and the Russian Federation (which has not yet ratified the protocol). The protocol fixes national annual emissions targets to be reached by 2010 for different gases: SO_2, NO_X, NH_3 and volatile organic compounds. It also imposes different practical measures, for the control of ammonia emissions from agricultural sources, to be taken by parties (with some qualifications related to technical and economical feasibility). These include an advisory code of good agricultural practice; solid manure incorporation within 24 hours of spreading; low-emission slurry application techniques; low-emission housing and slurry storage systems for large pig and poultry farms;[1] and prohibition of ammonium carbonate fertilizers and limits on ammonia emissions from urea.

The European Union adopted its own regulation on atmospheric pollutants: the 2001 National Emission Ceilings (NEC) Directive (directive 2001/81/EC of the European Parliament and of the Council). The NEC directive fixes national emission ceilings for the same gases, at the same level (except for Portugal) as the Gothenburg Protocol. The NEC directive is currently in the process of implementation. Member states had to build national programmes by October 2002, to be updated and revised as necessary in 2006, for the progressive reduction of their annual emissions.

Supporting intensification and promoting research and extension of cutting edge technology

If the projected future demand for livestock products is to be met, it is hard to see an alternative to intensification of livestock production. Indeed, the process of intensification must be accelerated if the use of additional land, water and other resources is to be avoided.

The principle means of limiting livestock's impact on the environment must be to reduce land requirements for livestock production, including the implicit water, nutrients and other resources represented by land. This involves the intensification of the most productive arable and grassland used to produce feed or pasture; and the retirement of marginally used land where this is socially acceptable and where other uses of such land, such as for environmental purposes, are in demand. The goal becomes more important where land for livestock production is marginal and its natural resource value is higher.

Intensification will lead to gradual reductions of resource use and waste emissions across the board. For example, precision feeding and use of improved genetics can greatly reduce emissions of gases (carbon dioxide, methane, etc.) and of nutrients per unit of output. Intensification in the form of a relative expansion of concentrate-based production systems, in particular chicken and other poultry, at the expense of ruminant production, in particular feed lots can reduce the overall impact of the livestock sector on climate change.

Intensification also needs to occur in the production of feedcrops, thereby limiting the use of land assigned to livestock production, either directly as pasture or indirectly for feedcrops. This will alleviate the pressure on habitats and associated biodiversity. While conventional intensification may increase the environmental burden on the areas involved, use of conservation agriculture (minimum tillage, precision use of water, fertilizers and pesticides, etc.) may mitigate this risk. Pasture intensification and improved feed cropping can sequester carbon, or at least reduce emissions of greenhouse gases.

Intensification needs to be brought about by price signals, corrected for current distortions

[1] More than 2 000 fattening pigs or 750 sows or 40 000 poultry.

and neglect of externalities, and will lead to a better utilization of natural resources used in the livestock production process, notably water.

As well as correcting input and output prices, public policies can play a facilitative role in intensification, by stimulating technology research and development. However, public technology research and development has considerably slowed down in the past decade (Byerlee *et al*., 2003). While continued research into productivity increases for commercial and industrial livestock and related feed production and use can be largely left to the private sector, public research needs to play a stronger role in natural resource management and in poverty reduction where accessible technologies offer such potential.

Purcell and Anderson (1997) analyse the role of research and extension and the role public policies can play in promoting these. They stress the importance of a conducive environment, including macro-economic and sectoral policies, favourable market opportunities, access to resources, input and credit. It is still widely considered that the amount of private research will always be less than socially optimal, and public stimulation of research must step in to fill the gap. In particular, this may apply to livestock-environment issues as public research and development needs to anticipate future scarcities. However, supporting public sector involvement in technology development will remain ineffective if the gross price distortions are not corrected.

Institutional development

While the livestock sector undergoes rapid transformation, institutions have lagged in responding to the environmental challenges that have arisen, for reasons discussed at the beginning of Chapter 4. Many resource degradation issues related to livestock are characterized by an absence of policies and institutions to address them.

Institutions are required to monitor environmental externalities, both negative and positive, and to ensure that these are accounted for and fed back into private decision-making. Institutions are also required to negotiate and sometimes implement these measures. Institutions are needed to develop standards and regulations and to enforce their implementation.

Institutional change is required to correct the policy distortions that currently create perverse incentives and encourage inefficient resource use and misallocation of resources. Very often, inappropriate price signals stem from lack of institutional capacity, such as, for example, in situations where traditional authorities have lost their grip over common property resources. Environmental stewardship needs to be established at the appropriate level: at communal watershed level in the case of common property grazing resources and water-harvesting schemes; at the national level for the protection of natural areas, for environmental policies and their implementation; at international level for the protection of the atmosphere and global issues related to biodiversity.

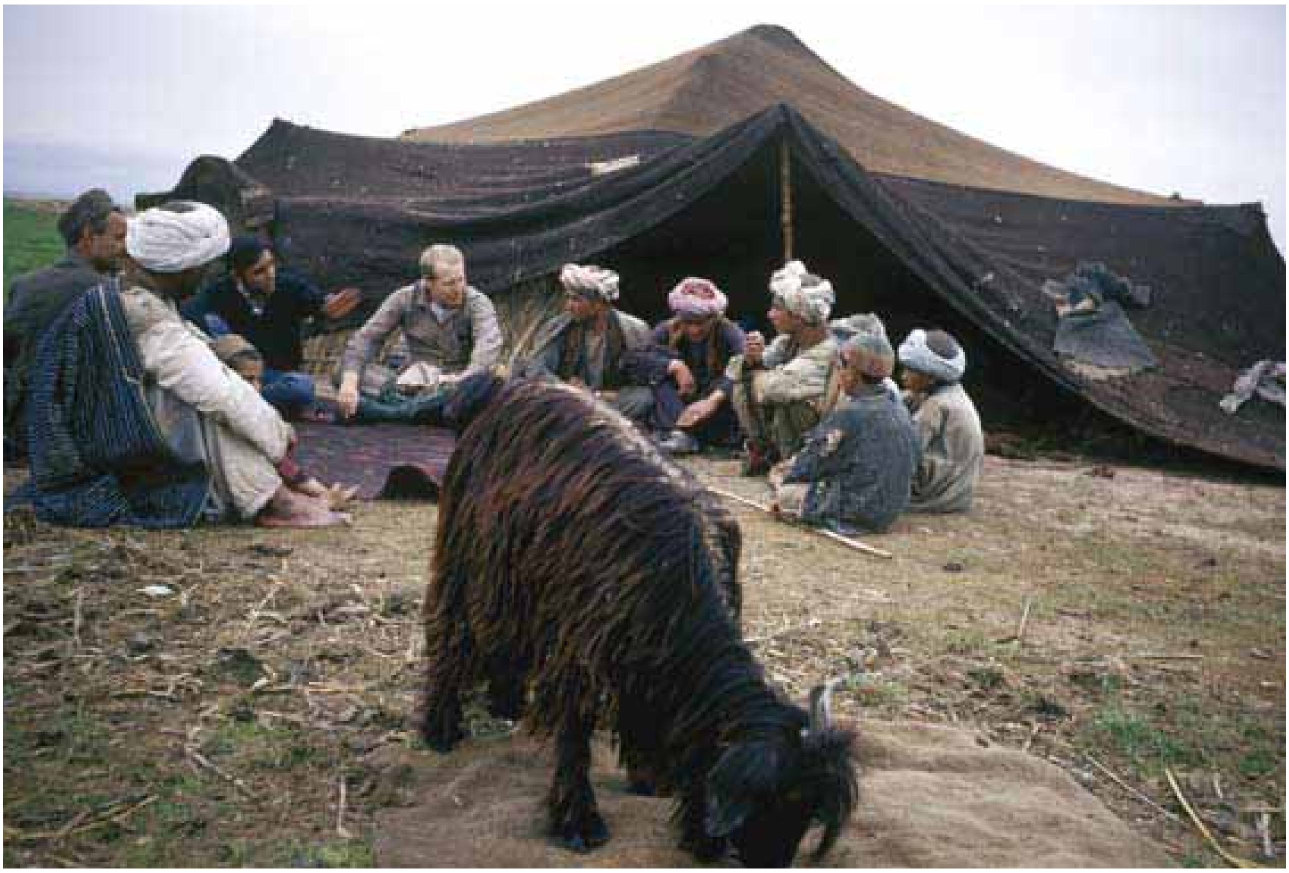
Awareness building, education and information

There is a pressing need to bring information about environmental concerns, and specifically awareness of the role of livestock in the degradation of natural resources, to the attention of the general public, of consumers, of pupils and students, of technical staff and extension workers, and of policy-makers and decision-takers in private business and public office. Communication among all stakeholders is important because most environmental issues related to livestock can only be successfully addressed in a concerted and negotiated way.

6.1.3 Policy issues in climate change

Having discussed general policy frameworks and approaches, we will look at their application in particular sectors beginning with climate change.

Agriculture (including livestock production) represents an important share of greenhouse gas emissions of many developing countries.



Project manager speaking with nomad shepherds in the north – Afghanistan 1969

However, it is apparent from the country emission reports submitted to the UNFCCC (National Reports, UNFCCC) that mitigation still tends to focus on other sectors. This is probably because of the technical difficulties related to assessing and certifying agricultural and land use, land-use change and forestry (LULUCF) sectors. However, progress is being made, and the potential contribution is huge.

Using the clean development mechanism

Currently the Kyoto Protocol's main mechanism for creating "certified emissions reductions" (CERs) that can subsequently be traded on the carbon market is the clean development mechanism (CDM). The CDM is a facility by which developed countries can reduce net carbon emissions by promoting renewable energy, energy efficiency or carbon sequestration projects in developing countries, receiving CERs in return. The purpose of the CDM is to help developed countries meet their obligations under the Kyoto Protocol while promoting sustainable development in developing countries.

The critical element for the success of the CDM is the participation of a broad cross-section of buyers (ultimately from developed countries) and sellers (from developing countries) of CERs. Three broad categories of projects qualify for the CDM:

- renewable energy projects that will be alternatives to fossil fuel projects;
- sequestration projects that offset greenhouse gas emissions (these are mostly in the LULUCF area); and
- energy efficient projects that will decrease the emissions of greenhouse gases.

For LULUCF projects, only afforestation or reforestation initiatives are recognized as being permissible during the Kyoto Protocol's first commitment period (2008-2012).

A critical factor concerning CDM transactions is an active international market for CERs which requires partnerships between several agents, namely project developers, investors, independent auditors, national authorities in host and recipient countries, and the international agencies that are responsible for implementation of the Kyoto Protocol (Mendis and Openshaw, 2004).

Since the protocol's ratification in February 2005, a considerable number of projects have been registered.[2] These projects are mostly based on predefined methodologies. Established methodologies in the livestock sector concern only emissions from the industrial production sector: the recovery of methane (as a renewable energy source); and greenhouse gas mitigation from improved animal waste management systems in confined animal feeding operations.[3] Scope exists for other types of projects aiming at mitigation of livestock emissions through intensification of production. For example, improving rumen fermentation efficiency through the use of better quality feed could substantially reduce emissions from the huge Indian dairy sector (Sirohi and Michaelowa, 2004). For this, credit (through e.g. micro-finance institutions), effective marketing, the use of incentives and promotional campaigns are required for broad acceptance of related technologies (Sirohi and Michaelowa, 2004).

Further problems relate to the fact that current CDM projects cannot be used to effectively alter a country's emission profile (Salter, 2004). A number of renewable energy projects would have major shortcomings, especially in terms of failure to demonstrate "additionality" and deliver added environmental and social benefits (Additionality refers to the situation where a project results in emission reductions over and above those that would have taken place - in the absence of the project). Defining what constitutes a baseline (the existing or projected greenhouse gas emissions in the absence of the project) is also problematic.

Seedlings being planted in an arid area for dune fixation. These activities form part of the rural forestry development project in the fight against desertification – Senegal 1999

Afforestation or reforestation (A/R) initiatives are the only land-use change projects that are currently eligible. However, they offer great potential for mitigating livestock's footprint on climate change by returning marginal, or degraded pastures, back to forest. Other potential methods that could significantly reduce emissions, but do not yet qualify for eligibility include: forms of pasture improvement, such as silvo-pastoral land use, reduced grazing pasture and technical improvements.

Promoting soil carbon sequestration

The effects of "leakage" may substantially raise the costs of carbon sequestration (Richards, 2004). "Leakage" occurs when the effects of a programme or project lead to a countervailing response beyond the boundary of the programme or project. This problem arises from two basic facts. First, land can be shifted back and forth between various forestry and agriculture uses. Second, the overall balance of activities on land will depend on the relative prices in the agriculture and forestry sectors. This is because individual projects and programmes do little to change prices or the resulting demand for land. For example, if forest land is preserved

[2] A list of registered projects and be found at http://cdm.unfccc.int/Projects/registered.html

[3] Methane recovery: http://cdm.unfccc.int/methodologies/DB/03E6PSPYME3LMKPM6QS6611K70A08F/view.html
Waste management: http://cdm.unfccc.int/methodologies/DB/3CQ19TPGO0FCG2XT08CP18P446L8SB/view.html

from harvest and conversion in one location, the unchanged demand for agricultural land and forest products could lead to increased forest clearing and conversion in another region. Thus, the effects of the preservation may be partially or entirely undone by the leakage. Similarly, if agricultural land is converted to forest stands, the underlying demand for agricultural land may simply cause other forested land to be converted back to agriculture.

Carbon sequestration programmes require different policy instruments than for carbon emissions control programmes (Richards, 2004). If carbon sequestration is either subsidized or used as an offset against carbon taxes or tradable allowances, then it will have quite a different effect on the system of public finance than an emissions control mechanism. In general, those instruments that require revenue-raising, such as subsidies and contracts, have a higher social cost than those that raise revenue, such as tradable allowances and emissions taxes.

Carbon sequestration activities require careful evaluation of the role to be played by government, to assess whether a pure market approach may be preferable to options under which the government retains more control over the type and manner of projects undertaken. One issue is the measurability and uncertainty of project outcomes. Another important point is the government's ability to credibly commit to maintaining incentives over long periods. Moreover, a carbon sequestration programme is likely to pursue multiple goals that may include erosion control, habitat provision, timber supply, and recreational enhancement. Thus, the goals of a sequestration programme are likely both to be difficult to measure and to shift over time. Similarly, Teixeira *et al*. (2006) suggest that a successful development of A/R projects in Brazil may require national policy involvement and regulatory action in addition to purely market oriented tools.

The potential for incremental accumulation of organic carbon in soils is huge and adapting extensive livestock systems is the key to unlocking this potential. Technical options to revert pasture degradation and sequester carbon, particularly in the soil by building up organic matter in the ground, exist and current pastures are probably the largest potential carbon sink available (see Chapter 3).

However, the same issues described above for A/R activities also apply here, e.g. "leakage", the pursuing of multiple goals, sustained government commitment, etc. The benefits accrue over a period of decades, in many cases peak carbon uptake rates occur only after 20-40 years. Landowners who make these investments will no doubt want to know whether the government will still be rewarding carbon sequestration long into the future when their activities come to fruition. Government needs to be able to make credible commitments to provide stable incentives over long periods.

While currently not eligible under the CDM, a most serious effort needs to be made to allow for certified emissions reductions from rehabilitation of degraded land and sustainable management of existing forest, be it under the CDM or in a different framework.

The potential benefits of improved soil carbon management are considerable and increase with scale. They include the:

- global level, climate change mitigation and enhanced biodiversity;
- national level, increased possibilities for tourism and enhanced agricultural sustainability and food supply; and
- local level, enhanced resource base for future generations and increased crop, timber and livestock yields (FAO, 2004b).

In the context of poorer developing countries, smallholders are a key group both in achieving the necessary scale, and in achieving developmental as well as environmental goals. In the absence of policy interventions and external financial support, smallholders use improved management practices at individually optimal levels but at socially suboptimal levels. On the basis of case studies, FAO (2004b) concludes that substantial funds from development organizations or carbon investors will be needed if soil carbon sequestration projects in dryland small-scale farming systems are to become a reality. The expected benefits are probably insufficient, without outside funding, to compensate farmers for costs occurring at the local level.

In addition to these purely economic calculations, there is an ethical concern. Expecting local smallholders to adopt management practices, at socially and globally optimal levels, implies that they subsidize the rest of society in their respective countries as well as global society. If sustainable agriculture, environmental restoration, and poverty alleviation are to be targeted simultaneously on a large-scale and over a longer period, then a more flexible and adaptive management and policy approach is needed. It should generate possibilities to strengthen farmers' own strategies for dealing with uncertainty while providing the necessary incentives.

Participatory approaches should be used. A long-term and large-scale carbon sequestration programme that might include several thousand individual smallholders is unlikely to succeed if all programme decisions are taken following an interventionist, top-down approach. This is likely to disillusion local farmers and increase the risk they will opt out of agreements. A first important step towards institutional integration is to identify already existing local and/or regional institutions that might be best suited to function as a vehicle for an anticipated carbon sequestration programme. In addition to being trusted by the majority of smallholders, such institutions should be able and willing to participate in the design of a local/regional programme; ensure the necessary participation of a large body of smallholders; guarantee a fair distribution of costs; coordinate monitoring and verification and channel eventual benefits in desirable and equitable ways (Tschakert and Tappan, 2004).

Soil carbon sequestration activities were not included as part of CDM in the first commitment period because of their complexity. However, they have great potential and they are among the goals of all major global environmental conventions - not only the Framework Convention on Climate Change, but also the Convention on Combating Desertification and the Convention on Biodiversity. There are a number of important alternative funding opportunities that could potentially be used to help implement carbon sequestration programmes: the BioCarbon Fund, the Global Environment Facility, the Adaptation Fund and the Prototype Carbon Fund (FAO, 2004b).

Substantial funds will be needed for soil carbon sequestration activities and the booming carbon or CER market may be a potential source. CER is one of the world's fastest-growing markets - some analysts project that it may be worth as much as US$40 billion dollars annually by the end of this decade . In 2004, the global volume of trade in CO_2 was only 94 million tonnes. In 2005, it rose to 800 million tonnes. In January 2006 alone, just among European players, the figure was more than 262 million tonnes for spot trading. When the Kyoto Protocol entered into force, a tonne of CO_2 sold for US$8-9 on the spot market. One year later, a tonne was changing hands at more than US$31.

6.1.4 Policy issues in water

Improving water efficiency is a critical objective as water resources become more scarce. From a technical viewpoint, improving the efficiency of water use refers to a reduction in losses. From an economic viewpoint it means increasing net returns to users while taking into account the externalities. Increasing water efficiency may mean some sectors give up water to other sectors where it has higher value uses. In some areas, this will lead to the preferential development of certain types of agricultural activities (Norton, 2003) and may reduce the output of the livestock sector.

Policies endeavoring to improve the efficiency of water use should focus on the adoption of appropriate water-efficient technologies, together with the management of water demand, in order to facilitate the use of water resources by the most water productive activities. This allocative efficiency can be achieved through the development of appropriate institutions governing water allocation, water rights, and water quality (Rosegrant, Cai and Cline, 2002). It is essential to include equity objectives in these policies, to distribute water equitably among the different actors so that no one will be deprived of access to this vital resource. Even if this objective is usually clearly mentioned in most policy frameworks, in reality it is often neglected (Norton, 2003).

Multiple policy instruments need to be included in water conservation policies. The appropriate mix of water policy instruments, water management reform and institutional arrangements have to be adapted to national and local conditions. Instruments will vary depending on the level of development, the agro-climatic conditions, the level of water scarcity, agricultural intensification and competition over access to water resources.

Voluntary participation should be the preferred strategy used; though coercion should be an available option (Napier, 2000). The implementation of adapted policy and technical options takes time, demands political commitment and finances (Rosegrant, Cai and Cline, 2002; Kallis and Butler, 2001).

Getting water pricing right

The fundamental role of prices is to help allocate resources among competing uses, users and time periods (Ward and Michelsen, 2002) and to encourage efficient use by users.

In practice, water for agriculture is, in many cases, provided free (representing a 100 percent subsidy) and even in countries where pricing systems have been instituted, water remains greatly under-priced (Norton, 2003). In many cases the introduction of water pricing, or attempts at reforming water prices, have stemmed from financial crisis, or pressure on government budgets, low recovery of costs, deteriorating infrastructure and increasing water demand (Bosworth *et al.*, 2002).

The general principles for water pricing have been set out by the Global Water Partnership (Rogers, Bhatia and Huber, 1998). In setting water prices, effluent charges, and incentives for pollution control, it is important to estimate the full cost of water used in a particular sector. This involves considering the following components (see Figure 6.2):

a) full supply cost (operation and maintenance and capital investment);
b) full economic cost (full supply costs plus the opportunity costs and economic externalities); and
c) full costs (full economic cost plus environmental externalities).

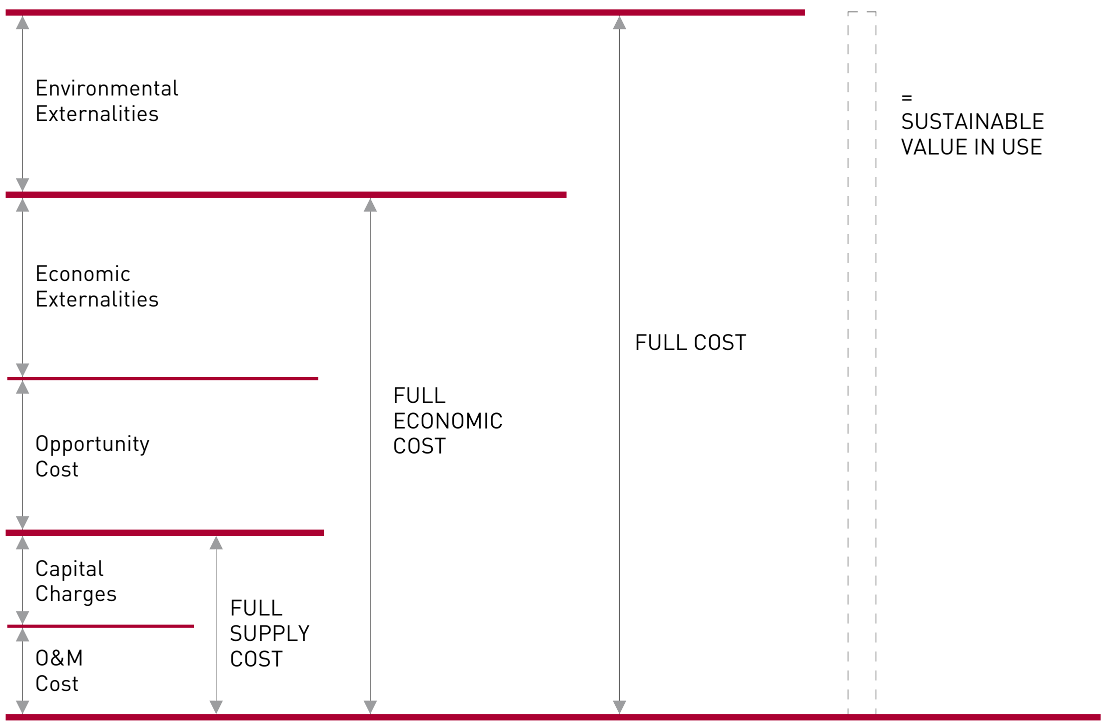
Prices should signal the true scarcity to users of water and the cost of providing the service; they should provide incentives for more efficient water use and provide service providers and investors with information on the real demand for any needed extension of water services. (Johansson, 2000; Bosworth *et al.*, 2002; Small and Carruthers, 1991).

Through measures, such as pollution charges and water pricing to encourage conservation and improved efficiency, pricing can serve as a means to ensure that actors internalize the environmental externalities that may arise from agricultural activities (Johansson, 2000; Bosworth *et al.*, 2002; Small and Carruthers, 1991). Adequate pricing can significantly reduce water withdrawals and consumption by agriculture, industry and households. Increasing water prices from the low levels prevailing in most countries can generate substantial water savings because of the high amount of water used in irrigation (Rosegrant, Cai and Cline, 2002).

Methods of water pricing

Water pricing methods include volumetric, non-volumetric, and market-based methods (Bosworth *et al.*, 2002; Johansson, 2000, Perry *et al.*, 1997; Small and Carruthers, 1991).

Figure 6.2 **General principles for pricing water**



Source: Rogers, Bhatia and Huber (1998).

Volumetric water pricing methods charge for water per unit of volume consumed. Volumetric water pricing is appropriate where the objective is to reduce water demand in the agricultural sector as well as reallocate water to other sectors. Volumetric methods depend on objective measurement of water abstraction and are often difficult to implement in practice. Several proxy methods or quasi-volumetric-pricing systems have been developed based on time of delivery, abstraction licences and block-rate/tiered volumetric methods.

Non-volumetric methods in agriculture can be based on agricultural outputs or area irrigated (Bosworth *et al.*, 2002; Johansson, 2000). These methods are usually used where the objective is cost recovery. Area-based pricing, where farmers pay a fixed price per unit of irrigated area, is the most common method of irrigation water pricing (Bosworth *et al.*, 2002).

In developing countries, the objective of water pricing is mainly to recover costs, more specifically operation and maintenance costs. In China for example, individuals are only charged for the pumping of irrigation water. However, the result is that only 28 percent of costs are recovered, providing little incentive to adopt water saving technologies (Jin and Young, 2003). In contrast, in developed countries the objectives are diverse and integrate demand management as well as the internalization of environmental externalities.

Water prices may consist of two components: a fixed charge and a variable charge. The fixed charge is intended to give the service provider a reliable stream of revenue, while the variable charge provides the user with the incentive to use water efficiently. The fixed component may be based on various denominators such as crop, unit area, duration of delivery, irrigation method or water velocity. The variable price component is based on the volume of water actually consumed (World Bank, 1997).

Not surprisingly, water prices tend to be higher in regions where water scarcity is an issue (Bosworth *et al.*, 2002). In countries such as Argentina, Bangladesh, India, Italy, Japan, Mexico, Pakistan, Spain, the Syrian Arab Republic, Sudan, Turkey, New Zealand agriculture is charged a flat rate based on the above denominators, whereas in Australia, France, Tunisia, the United Kingdom, the United States and Yemen users pay a varying tariff based on the amount of agricultural water consumed. At the other end of the spectrum, in Israel, farmers are given a water allocation for which they are charged on an increasing block tariff, according to the percentage of the allocation used. For the first 50 percent farmers are charged US\$0.18/m^3, for the next 30 percent US\$0.22/m^3 and for the last 20 percent US\$0.29/m^3 (Bosworth *et al.*, 2002).

A flat rate per hectare, based on the area irrigated or crop type - irrespective of the volume of water used - is unlikely to create any incentives for change. In a study on the effectiveness of pricing-based water policies in major irrigation districts in northern China (where water is charged at a flat rate on the basis of land area), Yang *et al.*, (2003) found that despite an increase in water charges, farmers' water-use did not change. Likewise, farmers in India and Pakistan and many other countries that pay area-based fees for water find their marginal cost of acquiring additional water to be zero - and therefore they have no incentive to economize on its use (Ahmed, 2000). Even where progressive block rates are being used, for example in Jordan, the progression of the prices and their levels are often too low to induce any change (Chohin-Koper, Rieu and Montginoul, 2003).

Handling difficulties in water pricing

Although volumetric methods represent an ideal approach to pricing of water, practical difficulties make them difficult to implement, especially

© FAO/19518/G. BIZZARRI

Water pump for irrigation – India 1997

in developing countries where farms are often small and scattered (Rosegrant, Cai and Cline, 2002). Problems include the objective measurement of water consumed plus transaction costs associated with monitoring and enforcement. As a consequence, proxies for volume of water are being used, such as length of delivery, the number of times a crop is irrigated and the share of a variable water supply to which a farmer is entitled.

The difficulty with volumetric pricing, at the level of the individual user, is sometimes overcome by a wholesale approach, whereby water is delivered and sold in bulk to organized groups of farmers at points where measurement of volume is feasible. Such water user associations consist either of farmers in smaller organization units that are common in Asia, or are specialized formal irrigation organizations such as those in Mexico and the United States (Hearne, 1999). Volumetric allocations are also common in Australia, Brazil, France, Madagascar and Spain (Bosworth *et al.*, 2002; World Bank, 1999; Ahmed, 2000; Asad *et al.*, 1999).

The fact that operation and maintenance costs are not, or not fully recovered, amounts to a subsidy for the crop and livestock sectors. Countries' experiences with cost recovery have been mixed. In a comparative study of 22 countries (World Bank, 1997), irrigation operation and maintenance cost recovery in developing countries has been found to range from a low of 20-30 percent in India and Pakistan (where the state remains heavily involved in the operation of irrigation systems) to a high of about 75 percent in Madagascar (where the role of the government is much reduced in favour of water users' associations who have been given responsibility for managing the irrigation systems). In OECD countries, the recovery of costs is much higher with the majority of countries obtaining full cost recovery for operation and maintenance cost. Countries like Australia, France, Japan, Spain, and the Netherlands also recover full supply costs from users (OECD, 1999). In the United States, state laws limit the charges that irrigation districts can impose on farmers to no more than their cost. Consequently, water prices are set to cover only costs of delivery and maintenance (Wahl, 1997).

The widespread under-pricing of water is a form of subsidy. These subsidies take several forms, including the public provision of water for agriculture at no or low prices, the subsidization of irrigation equipment or of energy for pumping groundwater. The removal of these subsidies is of prime importance in order to encourage efficient water use.

Agriculture generally enjoys subsidized water and is charged lower prices than industrial and domestic users. China, in pursuit of its objective of grain self-sufficiency, is stimulating grain production through the use of lower water charges for grain crops relative to other crops (Von Dörte Ehrensperger, 2004). In the United States, it was found that farmers pay as little as 1-5 cents per cubic metre while households pay 30-80 cents (Pimentel *et al.*, 2004). In Gujarat, India, electricity charges for groundwater pumping are subsidized - the charges paid by farmers for electricity are based on the capacity and not for the power used (Kumar and Singh, 2001). This amounts to a subsidy for water use and has contributed to water depletion and decline of the water table. Similarly, in France irrigation farming is on the increase, in part attributed to programmes that offer subsidies to farmers who invest in new irrigation equipment (OECD, 1999).

Subsidized development of boreholes in sub-Saharan Africa (mainly by development projects) has resulted in some places in the depletion of groundwater resources. In Namibia, for example, the provision of free water for livestock has resulted in water depletion, desertification and land degradation (Byers, 1997). Borehole development, the extensive use of groundwater coupled with the provision of water from canals and pipelines have been major contributing factors.

In many countries water pricing is a politically sensitive issue, especially where the economy is dependent on irrigation, as for example in China, Egypt or Sudan (Ahmed, 2000; Yang *et al.*, 2003; Von Dörte Ehrensperger, 2004). Moreover, an increase of water prices to a level that can influence behaviour may conflict with other policy objectives, including smallholder competitiveness, poverty reduction or food self-sufficiency. Furthermore holders of water rights may perceive the imposition or increase of water prices as an expropriation of those rights, thus reducing the value of their land (Rosegrant and Binswanger, 1994).

Creating the regulatory framework for water management

Regulations are often used to control pollution resulting from livestock activities or depletion of groundwater.

With water pollution, the establishment of water quality standards and control measures are central. While the use of uniform standards may simplify enforcement, smaller farms or enterprises may be unable to afford the costs of meeting the regulatory requirements or the waste treatment and relocation costs (FAO, 1999c). Hence standards can be defined locally or regionally, taking into account environmental and economic viewpoints as the marginal costs for technical adjustments may vary.

Regulatory mechanisms to control pollution can take a variety of forms:

- definition of minimum standards in order to reduce emissions and effluents to acceptable levels;
- specification of equipment to be used (effluent treatment) to meet the minimum standards;
- issuance of permits for the discharge of pollutants, which can also be traded. Tradable permits rely on payment-per-unit of pollution or the use of credits for reducing pollution. In that case market mechanisms are used to allocate pollution rights, once an acceptable overall level of pollution has been established; and
- specification of maximum industrial activity. For example, in livestock production systems limits may be placed on the number of livestock per hectare (FAO, 1999c).

These measures can be built into the codes that authorize access to water and regulate the water rights market (Norton, 2003). The establishment of penalties has to be done in a way that prevent their arbitrary removal by political edict. They should be of sufficient magnitude to act as effective disincentives to potential violators (Napier, 2000).

A set of criteria is used to monitor the impacts of livestock production systems on water quality and to set water quality standards for specific waterbodies. Parameters to be monitored to evaluate the impacts of livestock production systems include: sediment level; presence of nutrients (nitrogen, phosphorus and organic carbons); water temperature; dissolved oxygen level; pH level; pesticide levels; presence of heavy metals and drug residues; and levels of biological contaminants. The close monitoring of these parameters is a key element in evaluating compliance of production systems with defined standards and codes of practices. The European Commission proposes EU-wide emission controls and environmental quality standards for the substances and measures, its objective being the ultimate cessation, within 20 years, of emission of substances identified as hazardous (Kallis and Butler, 2001). Monitoring is costly and may represent a financial burden, especially in countries with limited monitoring capacities. Monitoring costs associated with the EU water framework directive was estimated at 350 million Euro for 1993 (Kallis and Butler, 2001).

Practices that pollute water resources are taxed in some places. For example in Belgium, wastewater from livestock production is either assimilated into domestic wastewater and taxed as such, or spread over agricultural land where it is subject to a special industrial tax (OECD, 1999). The EU water policy framework now includes a principle of "no direct discharge" to groundwater (Kallis and Butler, 2001).

Non-point source pollution is less easy to regulate. Codes of environmental practices and their enforcement are key elements in ensuring that agricultural activities that generate non-point source pollution would need prior authorization or registration based on binding rules (Kallis and Butler, 2001).

Extraction levels of groundwater resources are often regulated, especially in developed countries. Abstraction charges, especially within OECD countries, aim to control over-exploitation of groundwater resources. Countries where such charges are applied include Belgium, Bulgaria, Hungary, the Netherlands (Roth, 2001) and Jordan (Chohin-Koper *et al.*, 2003).

The extent to which groundwater protection policies have been effective is uncertain. Examples of policy failures are numerous, and users often have the opportunity to bypass environmental regulations. For example in the Netherlands, although farmers are subject to a groundwater extraction tax for water supplied for livestock production, they can extract the groundwater themselves without being taxed. In Belgium, while most livestock farmers pay wastewater tax, exemptions are given for about half of the water they consume (OECD, 1999).

Developing water rights and water markets

The lack of well-defined property rights in water often leads to unsustainable and inefficient resource use. In several countries, water rights are not defined and usually groundwater belongs to those who own the overlying piece of land. Hence, there is no restriction on the amount of water pumped by an individual land owner. In other countries, such as China, ownership of water is with the state, a fact that limits private incentives to conserve or use resources efficiently.

The proper functioning of water markets requires that water rights are formally and legally defined. In developing countries, such as Egypt, Pakistan and Sudan, water rights are insecure and poorly implemented with tail-end farmers often having insufficient water while farmers at the head take too much. Informal water markets, based on customary rights, are found for example in India, Mexico, and Pakistan. They usually consist of farmers selling surplus water to neighbouring farms or towns (Johansson, 2000). For example in Gujarat, India, rich landowners have invested in diesel pumps and pipe distribution networks to sell water to other farmers with no such equipment (Kumar and Singh, 2001). The development of a specific institution that manages the distribution and allocation of the rights may be required for conflict resolution mechanisms, for prevention of monopoly power and for the general enforcement of rules (Norton, 2003; Tsur and Dinar, 2002).

The organization of formal water markets is relatively new (Norton, 2003). The development of a water market will allow farmers to make decisions on whether to continue farming or to sell their water rights to the highest bidder and then improve water use efficiency. Australia, Chile, Mexico and the western United States are commonly cited examples of countries where formal markets and tradable water rights are being used to manage water allocation. Communal irrigation systems with tradable water rights are found for example in Nepal (Small and Carruthers, 1991).

Water markets show some pecularities compared to other markets. Usually transactions occur within the same watershed and even within the same irrigation system. Therefore, buyers and sellers are limited in number and the initial condition for a healthy market is generally not fulfilled. In northern Gujarat, India, informal groundwater markets are widely developed; although demand is lacking. Farmers are able to sell their excess of water to neighbouring farmers. However, efficient water allocation through these informal markets has not been achieved, because of the large number of sellers as opposed to buyers and the lack of opportunity to transfer water to other sectors.

Different types of water rights can be defined to fit with the market that will be established. Water rights should include a number of characteristics such as: the types of rights granted (total diversion rights, consumptive use rights or non-consumptive use), their duration, the system of sharing among users (ranked by level of priority among the users - appropriation system - or proportional rights among users) and the kind of users (rights can be delegated to individuals, private companies or communities) (Norton, 2003).

It is often hard to establish the initial water rights required by the system, because of the high costs related to water holding and capturing, and because the supply may be subject to unexpected changes (Ward and Michelsen, 2002). The allocation of free initial water rights, based on the existing use or right over access to water, can prevent conflicts associated with raising water prices and setting non-uniform charges. Furthermore, it can endow poor households with a valuable asset (Thobani, 1997 in Norton 2003, Rosegrant, *et al.*, 2002). Rosegrant *et al.* (2002) suggest that one solution to prevent conflict over the water price/water rights policy would be to apply a fixed base charge to an initial water rights baseline. For demand greater than the baseline an efficiency price equal to the value of water in alternative uses would be charged. On the other hand, for consumption below the base right, the water user would be paid back by the institution or the association (Rosegrant, Cai and Cline, 2002).

Paying for environmental services

Practices that lead to the provision of environmental services, such as improved water quantity and quality, can be encouraged through payments to the providers. Schemes of payment for environmental services (PES) rely on the development of a market for environmental services that have previously not been priced.

In a watershed context, upstream actors can be considered service providers if their actions result in improved water quality or quantity, for which they are compensated by downstream users. PES schemes require a market where the beneficiaries of these services (downstream water users) buy them from upstream providers. Obviously, this needs to be based on established cause-effect relations between the upstream land use and the downstream water resource conditions (FAO, 2004d).

PES schemes related to water services are usually of local importance at watershed level, with users and providers geographically close to each other. This facilitates the implementation of water-related PES schemes because of reduced transaction costs and easy information flow among the economic agents (FAO, 2004d), when compared to other types of environmental services with more remote or abstract linkages (carbon sequestration, biodiversity protection).

PES schemes are a promising mechanism for improving the condition of water resources in watersheds. They can sensitize the local population to the value of natural resources, and improve the efficiency of the use and allocation of these resources. PES schemes can also be used to resolve conflicts and can economically reward vulnerable sectors which offer environmental services (FAO, 2004d).

Nevertheless, the development of PES schemes is still at an early stage and implementation faces formidable difficulties. First, it is difficult to establish the relationship between land use and water-related services, as often the providers and users are not well identified. Usually, PES schemes rely on external financial resources; however, the long-term sustainability of the mechanisms is often uncertain. Furthermore, the level of payment is often politically imposed and does not correspond to effective demand for services (FAO, 2004d).

A few countries have specific legal frameworks for PES at the national or regional levels. Most of the existing PES schemes, however, operate without a specific legal framework. Some service providers take advantage of this legal gap to establish property rights for land and natural resources (FAO, 2004d).

The construction of large dams is usually associated with arrangements to reduce or eliminate grazing in water catchment areas that are susceptible to erosion and sedimentation. An example is the western China development strategy, attempting to reduce soil and water erosion and siltation into the Yellow and Yangtze rivers, which restricts or bans grazing in affected catchment areas, and in most cases provides compensation (Filson, 2001).

Coordinating institutional frameworks and participatory management

Implementation of better policies requires an adequate institutional framework. Typically, water resources are managed by several government ministries and departments (agriculture, energy, environment), which results in a fragmented decision-making process and lack of coordination among the different institutions (Norton, 2003). Water is a simple resource but its use is highly complex: different uses, by different users, controlled by different institutions in one part of the water cycle, may affect uses by other users in another part of the cycle. Both a strong coordination and an integrated approach involving all institutions are essential. Full cooperation between the different governmental bodies is a prerequisite to strategic planning and water policy implementation.

The development of specialized institutions is a key element in achieving the goals of a water agenda (Napier, 2000). The need to develop flexible and efficient institutions to maximize benefits from water use is obviously a pressing issue for economic development in dry places (Ward and Michelsen, 2002). The three main institutional approaches related to water policies are administrative allocation (public management), user-based allocation systems, and water markets.

Decentralization of the management of water resources and the involvement of user associations is another key aspect in the reform of existing institutional frameworks. The EU water framework directive is now following this approach. The implementation of its different policy measures will be coordinated at a "river-basin district" level. EU member states have designated river basin authorities within their own territories, and in coordination with other states for international waters (Kallis and Butler, 2001).

Institutional reliance on water user associations has proved to be effective. It improves local accountability, provides a mechanism for conflict resolution, and facilitates flexibility in water allocation. Furthermore, the costs related to information management for improved water resources allocation are significantly reduced (Rosegrant, Cai and Cline, 2002). In addition, recovery of operation and maintenance costs is improved. For example, in Mexico, a 30 to 80 percent increase in recovery rates. In Madagascar (where water users' associations manage irrigation systems) recovery rates are at the relatively high level of 75 percent (World Bank, 1997), because the responsibility of managing the irrigation systems has been transferred to the water users' associations. In contrast, where government continues to exert control over irrigation systems, as in China, India and Pakistan cost recovery is usually very low.

However, the transfer of responsibility for irrigation management to users will not necessarily ensure full cost recovery. Despite a definite increase in the levels of cost recovery, revenues are often still insufficient to cover full supply costs because water tariffs are generally set too low. The success of the transfer of irrigation management to water user associations is also dependent on the existence of a legal and institutional framework such as establishment of water rights.

Participatory watershed management is a key element in improving performance in water resources. Many watershed development projects have failed, or have performed poorly, because they did not sufficiently integrate and understand the local constraints and needs of local people (Johnson *et al.*, 2002). They suggested technology options that were ecologically and economically incompatible with local farming systems. Moreover, the new techniques imposed were exacerbating erosion as the new structures were not managed properly. Participatory watershed management programmes help local people define the issues, set priorities, select appropriate technologies and policy options adapted to their local context, and help sensitize them for monitoring and evaluation requirements (Johnson *et al.*, 20002).

6.1.5 Policy issues in biodiversity

While biodiversity loss is accelerating, the societal response to the problem has been slow and inadequate. This is caused by a general lack of awareness of the role of biodiversity, the failure of markets to reflect its value and its character as a public good (Loreau and Oteng-Yeboah, 2006). It has been suggested that an intergovernmental mechanism akin to the IPCC should be established, to link the scientific community to policy making, since the Convention on Biological Diversity is not in a position to mobilize scientific expertise to inform governments (Loreau and Oteng-Yeboah, 2006).

The area of biodiversity is intrinsically more complex than other environmental concerns and it is probably here that the gap between science and policy is largest. However, the scientific understanding of biodiversity and its functions has greatly improved in recent years, which is reflected in shifting attention from the side of policy makers. The scope of biodiversity conservation has been broadened to include protected areas and increased protection outside the designated areas based on the fact that whole ecosystems and their services often cannot be conserved by focusing on protected areas alone. New ways of financing biodiversity conservation are being explored to find alternative sources of funding. These include grants or payments from the private sector, conservation trust funds, resource extraction fees, user fees and debt-for-nature swaps at the governmental level.

A novel mechanism for conservation of biodiversity is the payment for environmental services approach, introduced in Section 6.1.4. Payments for environmental services are based on the principle that biodiversity provides a number of economically significant services. Payments need to be made to those who protect biodiversity to ensure the continued provision of these services. The environmental services that have received most attention are watershed protection and carbon sequestration. Other services, such as maintenance of biodiversity and landscape beauty, are also receiving increased attention (Le Quesne and McNally, 2004). Access charges and entrance fees to protected areas are also a form of payment for environmental services, in this case, conservation of biodiversity. They are not new, but recent schemes allow revenues to be used outside the protected areas and also be returned to local communities to provide incentives for biodiversity conservation (Le Quesne and McNally, 2004).

Recruiting land owners as protectors of biodiversity

A major challenge for new conservation approaches lies in the fact that in most countries endangered species are considered a public good, while their habitats are often on private land. As a private commodity, land can be transformed and traded. Biodiversity conservation can take place on private land but this relies on the owner's willingness and the land's opportunity cost. The opportunity cost of biodiversity conservation is difficult to estimate since the value of biodiversity depends on biological resources and ecosystem services.

The biological resources are not fully identified (the total number of species on earth is still unknown) and information on population numbers and risk status is still missing. However, some progress has been made in the valuation of ecosystem services. According to Boyd *et al.* (1999) the cost of conserving habitat should be valued at the difference between the value of land in its highest and best private use, and its value when employed in ways compatible with conservation.

To deal with the issue of ownership, new approaches have been tried, with relatively good success (Boyd, Caballero and Simpson, 1999). Most of these innovative approaches have been tried in forestry and at the community level, they can also be applied to livestock production.

- The purchase of full property interests involves the transfer of land from an owner who might develop the land to a conservator who will not. In order to purchase the property, the conservator must at least be able to pay the property owner the value of the land in private ownership. This value is the net present value of the land in whatever future use may be made of it, which is its opportunity cost. One of the distinguishing characteristics of full property interest acquisitions is that the conservator must compensate the landowner for the lost value of current financially productive land uses, as well as for the foregone opportunity of future conversion to more profitable use.
- Conservation easements are a contractual agreement between a landowner and a conservator. In exchange for payment (or as a donation that can be tax deductible) a land-

owner agrees to extinguish their rights to future land development. This agreement is monitored and enforced by the conservator, which may be a private conservation organization or governmental entity. Easements are often referred to as "partial interests" in land because they do not transfer the property itself to the conservator, merely the right to enforce prohibitions against future development.

- Another way to keep land out of development is for the government to give tax credits or other subsidies equal to the difference in value between developed and un-developed uses. For instance, if developed land earns US$100 more per acre than it does in low-intensity farming, a tax credit of US$100 per acre compensates the property owner for not developing the land. The subsidy is a cost borne by taxpayers.
- Tradable development rights imply a restriction on the amount of land that can be developed in a given area. Suppose, for instance, that the government seeks to restrict development by 50 percent in an area. It can do so by awarding each landowner the right to develop only 50 percent of their acreage. These development rights can then be traded. Tradable development rights impose costs on the landowners who have their development rights restricted. The aggregate opportunity cost is, as always, the value of development that is foregone in order to achieve the conservation goal. Though rights will be traded, the initial restriction of development opportunities imposes a cost on landowners. A tradable rights system has one particular advantage. Because property owners can, in effect, choose amongst themselves where development will ultimately be restricted, it leads to the least-cost development restrictions. In other words, development will be most restricted on those properties where the expected value of development is least.

Managing livestock and landscape for biodiversity conservation

Urban development causes major damage, stress and disturbance to ecosystems. McDonnell *et al.* (1997) studied ecosystem processes along an urban to rural gradient and found a cause and effect relationship, between the physical and chemical environment along the gradient and changes in forest community structure and ecosystem processes.

Livestock production is often structured along the urban to rural gradient, with industrial production systems in the peri-urban areas, feed-crops and mixed farming in rural areas, and extensive systems in the interface with wild habitats. This distribution, common in most countries, often places ruminant production in direct confrontation with wildlife and habitats.

In developed countries this interface is characterized mostly by wealthy or resource-rich farmers, operating under legislation for environmental protection, which is mostly enforced. In developing countries the interface is characterized by wide range stretching from resource-rich farmers to subsistence livestock holdings and herders. Even where legislation for environmental protection exists, it is often poorly enforced, or not at all. It is not surprising then that the major impact of livestock production is on habitat change. Land-use changes modify habitats extensively and are an important driver of biodiversity losses.

Prevention of perturbations is often the major goal of ecosystem management; however, disturbance is a natural component of ecosystems, and promotes biodiversity and renewal (Sheffer *et al.*, 2001). Ecosystems are subject to gradual and unpredictable natural events and respond by returning to their previous stable state or by shifting to an alternative stable state. Studies on ecosystem shifting (Sheffer *et al.*, 2001) suggest that strategies for sustainable management of ecosystems should focus on maintaining resilience enabling an ecosystem to absorb natural disturbances without crossing a threshold to a different structure or function.

The current state of thinking prefers landscape-focused conservation over site-focused conservation, particularly as an option to retain biodiversity in human dominated landscapes (Tabarelli and Gascon, 2005). Based on biodiversity conservation in corridors, the fundamental nature of landscape-focused conservation is to engage both conservation needs and economic development, by finding mutually beneficial interventions that might not necessarily occur within the buffer zones of protected areas. This may include new protected areas to protect watersheds, landscape management adding value to tourism, and the use of tradable development rights and easements to promote development compatible with the movement of species between protected areas (Sanderson *et al*., 2003).

Conservation efforts then should go beyond the protected areas and buffer zones to include a wide mosaic of land uses with a variety of production goals and socio-economic conditions of land users at the landscape level.

The integration of livestock production into landscape management poses many challenges for all policy and decision-makers and requires a truly holistic approach. The major challenges from the conservation point of view would be:

- to maintain the resilience of the ecosystem by predicting, monitoring and managing gradually changing variables affecting resilience such as land use, nutrient stocks, soil properties and biomass of long-term persistent species (including livestock); rather than merely to control fluctuations (Sheffer *et al*., 2001);
- to sustain the functionality of the ecosystem its capacity to sustain the processes required for maintaining itself, developing, and responding dynamically to constant occurring environmental changes (Ibisch, Jennings and Kreft, 2005). This includes the capacity of the ecosystem to provide environmental services; and
- to foster conservation efforts for taxa or species outside the protected areas, and to include forms of livestock development (land use and management practices) that are compatible with the requirements of such taxa or species.

To fully integrate livestock into landscape management it is necessary to recognize the multiple functions of livestock at landscape level. Apart from production objectives, livestock production can have environmental objectives (carbon sequestration, watershed protection), and social and cultural objectives (recreation, aesthetics and natural heritage) that should also be recognized, in order to achieve sustainable production. Livestock production has been proposed as a landscape management tool mostly for natural pasture habitats (Bernués *et al*., 2005; Gibon, 2005; Hadjigeorgiou *et al*., 2005) as it can constitute a cost-effective instrument to modulate the dynamics of vegetation to maintain landscapes in protected areas and to prevent forest fires (Bernués *et al*., 2005).

For an effective integration of livestock production into landscape management, radical changes should take place in management practices and land uses at the farm level. Recent research is focusing on new practices in managing grasslands, to address the relationships between grassland production and non-production functions. Among the research topics are:

- how management affects short- and long-term changes in grassland species composition and production - aiming to discover the impact of reduced fertilizer application on animal nutrition and N balance and/or the possibility of sustaining species-rich vegetation;
- the role of pasture vegetation, management practices and grazing behaviour on natural vegetation and faunal diversity, in both marginal and intensive livestock production areas, in relation to biodiversity conservation; and

- the spatial organization and dynamics of plant–animal grazing interactions at a variety of scales - with a view to optimizing the management of grazed landscapes so as to balance diversity, heterogeneity and agricultural performance; and
- The production and feeding value of species-rich grasslands - with a view to their integration in livestock production (Gibon, 2005).

However, the most important topic in relation to biodiversity conservation will be the issue of intensification because of its affect upon habitat change.

Agricultural intensification and land abandonment have considerable effects on biodiversity. In the EU, the decline of over 200 threatened plant species has been attributed to abandonment. Of the 195 bird species of European conservation concern, 40 are threatened by agricultural intensification and over 80 by agricultural abandonment (Hadjigeorgiou *et al.*, 2005). In grasslands it has been well documented that changes in vegetation patterns and structure that cause losses of biodiversity can result both from intensification of livestock production with increased use of organic and mineral fertilizers, and from intense grazing pressure without fertilization. Abandoned or low-grazed pastures, by contrast, result in encroachment of shrubby vegetation, causing losses of biodiversity and an increased risk of fires.

The issues of intensification and extensification will need to be managed at the landscape level according to socio-economic and environmental conditions. The optimal approach will probably be a mixture of intensification on land area, extensive grazing and setting aside land for conservation structured along the gradient: farm - communal area - buffer zone - protected area.

The driving factors that should be addressed at the landscape level are degradation and shrinkage of common land, high livestock densities, lack of common property management and inequity in the distribution of watershed benefits. Intensification of livestock production can contribute to biodiversity conservation at the watershed level. This would include pasture development, multipurpose trees for fodder, fuel or timber and improvement of the genetic capacity of local breeds. It would be accompanied by payments for environmental services (biodiversity protection, carbon sequestration and water quality) and a rationing system for common property resources (e.g. grazing fees).

From the point of view of biodiversity conservation, perhaps the major challenge in incorporating livestock into landscape management is to integrate livestock producers into conservation efforts at the landscape level. From the land users' perspective, biodiversity conservation is often considered as an externality, as are the improvement of water quality and availability and carbon sequestration benefits. As such, land users do not take them into consideration in making their land-use decisions, thus reducing the likelihood that they will adopt practices that generate such benefits.

Biodiversity conservation also implies the preservation of species that may hinder livestock production. In Latin America for example, poisonous snakes and vampire bats are considered agricultural pests for cattle rearing - they are considered as biodiversity instead of biodiversity. Under landscape management, farmers should incorporate conservation goals into livestock production. This will entail diversification of production; adoption of good management practices such as reduction of fire, pesticides and mineral fertilizers; and maintenance of the functional connectivity between livestock and the wildlife uses through different land uses at the farm and landscape level. There are many technical possibilities for maintaining functional connectivity on farms. They include live fences, biological corridors, land set aside for conservation, protected areas inside farms and fencing of riparian forests. At the landscape level functional connectivity can be enhanced by wildlife corridors to connect protected areas and isolated patches of forests.

Policies are needed to guide the current opportunistic development process of livestock development at the landscape level for preservation of biodiversity. One of the main issues for the formulation of policies is that at landscape level, property boundaries do not correspond with ecological boundaries. The number of land owners and the mixed set of ownership types (public and private) ensure that individual owners' decisions have an affect upon the decisions of neighbouring land owners (Perrings and Touza-Montero, 2004). Enforcement, auditing and monitoring mechanisms and decision support tools should be embedded into the policy framework.

Regional policy trends and options for management of livestock/biodiversity interactions

In the **European Union** the current trend in grasslands is towards more extensive use of pastures, particularly in valuable ecosystems. Driven, among other things, by the need to reduce agricultural surpluses, by pressures from social concerns about animal welfare and by consumer preferences for organic farming, the EU Agri-environmental Regulation, in place since 1992, sets limits on application of fertilizers to grasslands and offers incentives for extensive use of sensitive areas and the maintenance of biodiversity and landscapes (Gibon, 2005).

In **Latin America**, where the deforestation of biodiversity-rich habitats is linked to extensive livestock production, intensification of land use should be a priority, through the use of pasture-legume mixtures or silvopastoral systems, combined with incentives for setting aside land for conservation, delineation of sensitive areas, payments for environmental services such as carbon sequestration and biodiversity conservation.

Africa is a mosaic ranging from well-developed landscapes to relatively unchanged habitats, with a wide diversity of land uses and interactions with biodiversity. A major impact of the changing landscape has been increasing competition for the finite resources among growing human populations, many of them desperately poor. As a consequence, the wildlife/livestock interface has become more conflicted in certain areas of Africa, although in others it is no longer an issue (Kock, 2005). In arid and semi-arid lands where wildlife, livestock and people interactions are intense, arable agriculture has expanded into marginal lands and open communal grazing lands (Mizutani *et al.*, 2005).

There is growing evidence that both cattle ranching and pastoralism can have positive impacts on biodiversity. Ranching can do so by intensification and consequent reduction of herd size, along with sustainable exploitation of wildlife resources. Pastoralism can do so by adjusting grazing patterns so as to provide dispersal zones for wildlife outside the protected areas (Kock, 2005). The challenge, at the landscape level, is to match land use with ecological processes, so as to exploit the temporal and spatial variation of key resources to allow wildlife and livestock production (Cumming, 2005). African grasslands in humid and subhumid zones are subject to strong economic incentives to develop intensive ranching and agriculture, mostly at the expense of wildlife. The reason is the large difference in profits and revenues between traditional livestock management and using the land to its full agricultural potential. From the viewpoint of biodiversity, extensification will bring the best opportunities for conservation; however this needs the right mix of regulations and incentives to find acceptance. Tradable development rights and conservation easement schemes may be required to compensate landowners for not developing their land (Norton-Griffiths, 1995).

In the grasslands of the **Commonwealth of Independent States**, problems have arisen of intensification close to villages in pastoral areas and of land abandonment in remote pastures. These linked problems derive from widespread poverty along with several trends in the livestock sector:

- concentration of animals in peri-urban environments;
- disruption of transhumance herding by official sedentarization policies and other factors;

- lack of infrastructure and access to markets in remote pastures;
- lack of appropriate technology for pasture management; and
- fragmentation and change in composition of livestock holdings.

Land leasing is currently too cheap and this does not encourage the livestock farmers to take care of the land and to move to more distant pastures. On the other hand, livestock keepers in remote pastures do not have access to services, and are not compensated for the environmental services they provide.

A key strategy to encourage pastoralists to move away from pastures near villages, back to remote pastures, may be the creation of a pasture fund based on revenues from land leasing, with additional support from payments for environmental services, especially carbon sequestration. The pasture fund could have differential leasing prices - higher near villages and lower in remote pastures. It could also reward livestock farmers who make sustainable use of the land and introduce good management practices, by reducing the leasing prices, while fining farmers who do the opposite, by increasing their leasing prices. The pasture fund would also support transhumance by providing livestock services along the migration routes. A small increase in taxes on water would generate additional revenues to support the pasture fund, given that livestock farmers help to sustain water services especially in hilly and mountainous areas (Rosales and Livinets, 2005).

In the semi-arid and arid-lands of **India**, livestock production plays a crucial role in the management and utilization of fragile ecosystems. Under these conditions, animal husbandry is the traditional and major source of livelihoods, while arable farming plays more of a complementary role. However, growing human and livestock populations, and the adoption of non-sustainable practices, have lead to a rapid depletion of natural resources (especially of common property), which is affecting the functions of entire watershed ecosystems. Reduced availability of natural resources has already seriously affected the poor, marginalized and landless people, especially women, who depend on these resources for maintenance of their livestock and their own livelihood.

Integrating protected areas and livestock management

Since 1950, areas designated as protected by national legislations have been growing at a fast pace all over the world (see Chapter 5). Despite this, the number of species at risk of extinction and the destruction of habitats have also risen. At the same time, livestock numbers have increased at a steady rate along with the growth of human populations. There is an urgent need to change livestock production and conservation approaches to lessen the impacts on biodiversity.

Current conservation efforts have been criticized for focusing on single species rather than on ecosystem functionality (Ibisch, Jennings and Kreft, 2005). Protected areas can be effective for pure conservation purposes. Although, their effectiveness in providing and maintaining a full range of ecosystem services is often very limited, since many protected areas are too small and spatially isolated (Pagiola, von Ritter and Bishop, 2004). Protected areas also suffer from inadequate legislation and management, lack of resources and insufficient stakeholder involvement (MEA, 2005b).

Where the primary objective of protected areas is to maximize conservation, the primary objective of livestock production is to maximize productivity and earnings. Experience shows that these two objectives are often mutually exclusive. Most of the conflict could be alleviated if the goals of livestock production were broadened to include ecosystem conservation, services and management, rather than only to produce food. Conflict would also be alleviated if biodiversity conservation goals were broadened to include preservation outside the protected areas while maintaining the functionality of natural ecosystems in an integrated mosaic with food production at the landscape level.

Service oriented grazing

Livestock production is an important source of foreign currency, providing over half of the value of global agricultural output and one-third in developing countries. It is also a key element in the fight against poverty as approximately one-quarter of the global poor (of whom 2.8 billion live on less than US$2 per day) are livestock keepers.

PES offer a way of combating poverty and simultaneously addressing many other critical socio-economic and environmental goals by:

- integrating livestock production, particularly of ruminants, with conservation goals;
- using livestock as a tool for landscape management; and
- recognizing the benefits of biodiversity conservation and carbon sequestration.

PES have been discussed in the preceding sections. In the case of biodiversity such schemes are more difficult because of difficulties in measuring and valuing biodiversity. However, the MEA (2005) shows that protected areas function best when benefits from biodiversity preservation can be captured by local people.

6.2 Policy options for addressing environmental pressure points

6.2.1 Controlling expansion into natural ecosystems

The expansion of pasture areas into natural ecosystems has essentially come to an end in most parts of the world, except for Latin America (in particular the central part of South America) and central Africa. In Latin America, many currently forested areas are attractive for cattle ranching. Indeed, currently 70 percent of previously forested land in the Amazon is occupied by pastures. This has important consequences for humid tropical ecosystems. In contrast, the presence of trypanosomiasis in the humid and subhumid parts of Africa continues to constrain a similar expansion. Here, arable land (such as shifting cultivation or fallow cultivation) is the predominant land use following deforestation. Only when the habitat has become unsuitable for the vector of trypanosomiasis, the tse-tse fly (*Glossina spp.*), as a result of human population increase and expansion of cropping, can grazing animals move into the cleared areas.

The main policy issue, with regard to pasture expansion and related deforestation, lies with land titling and land markets, and with the weaknesses in establishing and enforcing regulations in remote areas such as the Amazon. Here, livestock are often used as a tool to occupy land for speculative purposes. At the initial, speculative phase of deforestation, forests are cut down or burned and occupied with cattle, on the expectation that land titling will be granted at a later point on the basis of such occupancy. In these situations the incentive for efficient land use and good land management is weaker, and livestock-induced degradation is more likely to occur. Land titling, and related institutional capacity, need to be quickly expanded and upgraded to stem the loss of valuable resources.

However, deforestation for cattle ranching has proven to be profitable in itself, from a micro-

Table 6.1

Comparison of key technical parameters in the beef industry in the Amazon area of Brazil (1985-2003)

	1985	2003
Carrying capacity (AU/ha)	0.2–1	0.91
Fertility rate %	50–60	88
Calf Mortality %	15–20	3
Daily weight gain kg	0.30	0.45

Note: AU=Animal Unit is a standard to aggregate different classes of livestock, with adult bulls at 1 AU, cows at 0.7 AU , yearlings at 0.5 AU and calves at 0.2 AU.
Source: Margulis, 2004. Data from the entire North West Brazil in World Bank 1991 Brazil: Key Policy Issues in the Livestock Sector-Towards a Framework for Efficient and Sustainable Growth" Agricultural Operations Division, Report no 8570-BR Washington DC

economic perspective, in areas where titling is consolidated (Margulis, 2004). This, in large part, is the result of major improvements in the technology used in cattle ranching that have occurred in past years as shown by Table 6.1.

Land speculation also plays a role. The fact that land is still, in some parts of the world, unreasonably cheap, encourages horizontal expansion and extensive use of such land, in particular in the humid tropics of Latin America. Driving up the cost of holding land, by making squatting more difficult, and by taxing land ownership (perhaps with a tax-free minimum) will encourage productivity increases and enhance environmental sustainability. Land taxes have shown considerable potential to drive land use towards higher productivity, thereby limiting its use for speculative purposes. The introduction of deforestation taxes also appears to be a suitable instrument if they can be imposed (Margulis, 2004).

Zoning can be an effective instrument if there are functioning institutional frameworks to assign and police land uses. In the case of valuable natural resources associated with land, creation of protected areas may often be the preferred strategy. Zoning may also include limits on the number and size of livestock permitted, based on the vulnerability of the land to soil degradation and erosion (FAO, 2006). However, because of weak institutions in most areas concerned, usually remote areas in developing countries, there are problems with enforcement of zoning and encroachment on protected areas. To improve compliance, land policies and rules need to be developed in harmony with the interests and needs of pastoralists and other livestock owners. However, as Margulis (2004) indicates, in view of its enhanced commercial attractiveness it will be difficult to stop the expansion of ranching altogether, but it could be directed towards less valuable ecosystems, thereby saving those that are of most value.

Infrastructure policies also play a role. As the presence of infrastructure, and the expectation of future infrastructure development, has been identified as a powerful determinant for land use (including conversion of forests into pastures), infrastructure development planning needs to take this into account. Caution should be exercised so as to open areas only when there are functioning authorities to control access, land titling, area protection and law enforcement.

Public research and extension can help in driving land use towards more productive and sustainable forms, by developing technical packages focusing on intensification, including improved pasture, intensified dairy or beef production and the inclusion of forests and silvo-pastoral land use on farms. Research (Murgueitio, 2004; Olea, Lopez-Bellido and Poblaciones, 2004) has shown that such forms of land use are profitable, particularly for small farms with a relative abundance of labour, and can generate significant environmental benefits.

An associated issue is the degradation of pasture in previously forested areas. A large part of tropical pastures (estimates range up to 50 percent) are seriously degraded, caused by unsuitable terrain (slopes), and high rainfall. Deforestation and the spontaneous establishment of pastures without any protective measures or improvements, leaves the soil exposed and subject to erosion. The ensuing degradation can be addressed by forms of silvo-pastoral land use that mimic the original vegetation to a certain extent (see Box 6.2).

PES schemes have the potential to provide incentives for land-use change; the problem is to make such schemes sustainable so that change becomes permanent. The most immediate option lies in payment for water services, as benefits in improved water flows and quality would directly benefit local communities downstream. Silvo-pastoral systems, in combination with other measures of water protection, considerably reduce runoff and sedimentation of reservoirs. Payments for carbon sequestration are another option, which will depend on the development of effective carbon markets (see

Box 6.2 Payment for environmental services in Central America

The Global Environmental Facility (GEF) and the World Bank support a regional project in Central America, which uses payment for environmental services, as a tool to promote the conversion of degraded pastures towards more complex vegetations, which increase carbon sequestration and enhance bio-diversity. The adopted methodology was designed to reduce transaction costs[1].

- Different vegetation units were ranked by an expert panel, on their contribution to carbon sequestration and bio-diversity;
- Using satellite technology, an inventory of the main vegetation units was made of each farm. On the basis of this inventory a baseline was established;
- Each year, changes in the different vegetation types were measured, and used as a proxy for the payment. The level of payment was based on the equivalent of US$5 per tonne of carbon. In the absence of a functioning market for bio-diversity, about the same level was, rather arbitrarily, set for this aspect; and
- The project design features supported the simplicity: Payment was on the basis of performance (ex-post), the farmers had to obtain their own sources of funding, thus avoiding complex rural credit schemes, all funding was channeled through NGOs.

About 200 farmers in six watersheds in three countries (Colombia, Costa Rica and Nicaragua) participate in this scheme. The results, after three years of operation are promising:

- The relationship between vegetation types and carbon sequestration and biodiversity enhancement was strong, showing that vegetation types can be used as a proxy for the measurement of environmental services;
- Ranchers reacted very positively to the incentives provided. A total of about 2 000 hectares were established with improved, deeper-rooting pastures and more trees, more than 850 km of living fence were established, which significantly improved the connectivity of the different habitats, and about 100 hectares in slopes were left in fallow to regenerate to secondary forest. The average payment per farm was about US$38/ha in the second year of operation; the average monitoring costs about US$4/ha;
- Poorer farmers found the resources for the required investments. A survey found that the poorer farmers received higher payments per hectares than the larger ranches; and
- The reaction of the public institutions was quite favourable. In Costa Rica, the government decided to include agroforestry (and this scheme) in its forest environmental service payment scheme, which is funded through fuel taxes and water charges. In Colombia the National Livestock Federation is negotiating international and national funding sources to up-scale this pilot operation.

The biggest challenge will be to further simplify the methodology and find the international funding sources, linked to carbon trading, which will enable the application of such payment schemes for areas such as the Amazon, to tip the balance from continuing expansion to intensification of production.

[1] See also FAO (2006) (available at www.fao.org/AG/AGAINFO/resources/documents/pol-briefs/03/EN/AGA04_EN_05.pdf).

Source: Pagiola, von Ritter and Bishop (2004).

Section 6.1.3). In some cases, new opportunities for payment schemes are arising, such as for Costa Rica, where part of the fuel tax is used for such purposes. Payments for biodiversity protection are, at present, mainly in the form of tourism revenues.

6.2.2 Limiting rangeland degradation

The expansion of pastures into natural habitats over the last two centuries has been driven by the quest for additional food and other resources for growing populations. As described in Chapter 2, when introducing the concept of the livestock transition, pasture expansion has reached its peak in most parts of the world, occupying areas that are, at best, marginally productive, which are, in many ways, unsuitable for sustained production. Growing demands for environmental services are starting to compete with traditional forms of low-output livestock production, leading to progressive abandonment of marginal pastures.

Degradation of rangeland, on both communal and private lands, is a pressing issue in many countries, including developed countries. Degradation of rangeland has important negative consequences for water resources and biodiversity and is an important source of greenhouse gases. These problems are particularly pronounced in areas where the livelihoods of many poor people depend on livestock, and on the common pasture that sustains them, and where alternative livelihood options (such as urban employment) are absent. These conditions are widespread in arid and semi-arid zones of sub-Saharan Africa, and parts of the Near East, South Asia and Central Asia (see Map 26, Annex 1).

Under common property regimes, overgrazing of common property resources is often caused by mobility restrictions. These arise from the expansion of rainfed cropping in key dry-season grazing areas for mobile systems, land privatization, fencing and establishment of irrigation schemes. Pastoralists require improved access management to pasture resources, including

© FAO/19428/R. FAIDUTTI

Spontaneous regeneration of mountain vegetation after a four year ban on grazing and cutting down trees – 1996

regulations controlling grazing and stocking rates. A key characteristic of the dry areas is the extreme variability of the rainfall, and hence bio-mass production. Fixing livestock numbers under such extreme variability is, therefore, counterproductive. What is needed are strong institutions and infrastructure, in particular for livestock marketing, which can adapt livestock numbers to the prevailing climatic conditions and standing biomass. Therefore, grazing management becomes risk management.

However, to counter the degradation of common property resources, in particular grazing land, overall grazing pressure needs to be lowered. However, this is difficult to implement under common property regimes in the absence of a strong local, traditional or modern, authority. Because of the increasing fragility of traditional institutions in developing countries frequently a mix of traditional and modern authorities is needed to achieve the type of collective action required.

In many cases, compensation schemes are needed, or payment-for-services schemes where herders receive payments for improved water management, which benefits water supply or reduces siltation of dams. Similar forms of payment schemes, including benefit sharing, have been developed to facilitate the harmoni-

Box 6.3 **Wildlife management areas and land-use planning in the United Republic of Tanzania**

Pastoralism is the dominant land use and livelihood strategy in northern Tanzania, one of the world's richest remaining refuges for wildlife. If properly managed, nomadic pastoral livestock production is potentially the most environmentally compatible agricultural activity in this ecosystem.

One of the main threats to biodiversity in pastoral ecosystems is the breakdown of traditional adaptive and flexible management strategies developed by pastoral communities to optimize the use of temporally and spatially variable natural resources. The spontaneous spread of agriculture throughout this semi-arid ecosystem, by both settled pastoralists and external agents, has resulted in habitat change and truncation of important ecosystems.

If returns from wildlife could be shared with pastoral households this could stem the expansion of crop cultivation. Currently, pastoralists bear most of the costs of wildlife in the form of predation and competition for grazing and water, but do not gain any of the potential substantial benefits. What is required is the integration of sound wildlife management with wildlife-compatible land use by pastoralists.

The Government of the United Republic of Tanzania has established a series of policies to improve the distribution of the benefits generated by wildlife to affected communities and to carefully plan the use of the common resources to protect the interest of the three main stakeholders i.e.: wildlife, croppers and herders. In this regard, the wildlife policy established in Tanzania in 1998 called for the creation of wildlife management areas (WMAs). WMAs give local communities some control over wildlife resources on their lands and enable them to benefit directly from these resources. When WMA are established, communities may lease trophy hunting or game viewing concessions to tourism operators or engage themselves in hunting. At the same time, the WMA policy, the National Land Policy and Land Act (1999) and Village Land Act (1999) promote village land-use plans to ensure the appropriate management of communal land.

The LEAD-GEF project entitled "Novel forms of livestock and wildlife integration adjacent to protected areas in Africa" is supporting the evolution of community-based natural resource management in Tanzania. This project implemented in six villages in the Simanjiro and Monduli districts includes the development and implementation of participatory land-use planning and WMAs; the design and the implementation of benefit sharing mechanisms to increase returns from integrated wildlife and livestock production systems including the development of conservation business ventures with private partners; and the development of decision support tools in order to strengthen sustainable resource access and management.

Source: FAO (2003c).

ous co-existence of wildlife and livestock in sub-Saharan Africa, some of which have been pioneered by the LEAD-Initiative (see Box 6.3).

Maintaining animals on communal land is economically attractive even if returns are low as long as costs are minimal; this results in overstocking. If priced appropriately, grazing fees and other forms of costs related to the number or units of animal grazed on communal grazing land will encourage herders to limit grazing pressure, by taking out unproductive animals and by de-stocking early. For example, such a grazing fee is common practice in Morocco. Such grazing fees could also be progressive, with higher fees paid for larger herds. Similarly, making grazing rights tradable could establish market mechanisms for resource use, which is particularly important when pastures are under temporary (drought) or permanent pressure. While these are potentially viable options, control and enforcement is a common problem.

Mobility is a key management requirement in many arid areas with highly variable rainfall, and limitations of mobility have been identified as a key determinant in resource degradation (Behnke, 1997), because they concentrate grazing pressure over-proportionally in certain areas. Where such limitations exist, institutional arrangements must be found for passage agreements to allow pastoralists to balance out grazing resources. This is becoming increasingly difficult as both rainfed and irrigated agriculture encroach into previous pastoralists' areas. Public institutions have a role to play in helping herders de-stock early in the case of drought, if necessary also in the form of market interventions. Early destocking can reduce environmental damage and vegetation recovers more quickly when the drought is over. Subsidies that would enable early destocking have been used in some places, such as in Morocco.

In high-income countries, and where there is widespread degradation of state-owned land leased out to individual farmers, such as in western Australia or in the western United States, there is a strong pressure to convert these marginal lands back to their original state. In the light of the small contribution that these areas make to overall livestock supply, and the growing demands for other uses such as recreation or environmental services for these areas, this is a real possibility in the long term.

While important to the livelihoods of millions of pastoralists and ranchers, extensive grazing areas occupy immense lands with sometimes devastating environmental consequences, but contribute little to overall food supply. With growing resource pressure and demand for environmental services, there will be increasing pressure to take these areas out of production. It will fall upon public policies to develop a way out for the people concerned, and to find alternative income and employment outside the extensive livestock sector. For those who remain, practices need to change in line with the growing and differentiating demand for these land resources hitherto considered of little value. The potential of dry lands to provide environmental services such as water protection, biodiversity conservation and carbon sequestration will easily offset the values currently generated through livestock production, if effective markets can operate.

Water is a critical resource in extensive livestock production, and is often supplied through public infrastructure and without charge, under policies that are driven mainly by social considerations. Yet often the infrastructure cannot be maintained. Cost recovery for water provision and forms of more appropriate water pricing will allow maintenance and improvement of infrastructure, and will also lead to more efficient water use, and better allocation of water among competing agricultural and non-agricultural uses. Full cost recovery needs to be applied, both for grazing under common property regimes and for private ownership.

Resource costs, price distortions and externalities vary among livestock products. Beef has been identified as carrying the largest costs in terms of land and water requirements for its production, as well as in terms of contribution to climate change. It can, therefore, be argued that relative to other forms of animal protein, beef carries the largest externalities and benefits most from price distortions. Since immediate changes in land and water prices for its production may be difficult to implement, governments may consider the option of taxing beef. Demand for beef would then decline relative to other meats, and the pressure on both extensive grazing resources and feedgrain areas would be reduced.

6.2.3 Reducing nutrient loading in livestock concentration areas

Another facet of the livestock transition is the ongoing concentration of livestock in specific favoured locations, such as those offering easy access to urban markets, or close to feed supplies. The separation of livestock production and the growing of feed crops is a defining characteristic of the industrialization of livestock production (Naylor *et al.*, 2005).

Nutrient loading is caused by high animal densities, particularly on the periphery of cities, and by inadequate animal waste treatment. Issues of nutrient loading are present in developed countries, but they are particularly pronounced in emerging economies with rapid industrialization of the livestock sector, such as Brazil, China, Mexico, the Philippines and Thailand. Map 4.1 (Chapter 4) gives a regional overview of areas facing such nutrient loading for Asia. Other affected areas mainly include coastal areas in Europe, Latin America and North America; also some inland areas such as parts of Brazil and the midwest of the United States.

Major forms of pollution, associated with manure management in intensive livestock production, were described in Chapter 4. They include (FAO, 2005e):

- eutrophication of surface water, killing fish and other aquatic life;
- leaching of nitrates and pathogens into groundwater, threatening drinking-water supplies;
- build up of excess nutrients and heavy metals in the soil, damaging soil fertility;
- contamination of soil and water resources with pathogens; and
- release of ammonia, methane and other gases into the air.

Policies to address the issue of nutrient loading include instruments to influence the spatial distribution of livestock, so as to avoid excess concentration, reduce waste per unit of output, by increasing production efficiency and regulation of waste management (FAO, 2005e).

The LEAD-Initiative has conducted a variety of studies and programmes (Tran Thi Dan, 2003) targeted at better geographic distribution, in what has been called area-wide integration of specialized crop and livestock activities. These efforts aim to re-connect nutrient flows from crop and livestock activities in a watershed context, for example by recycling manure on cropland, as these activities become increasingly disconnected with specialization and economies of scale. This takes into account that, where economic pressure makes family-based mixed farming unviable, one should still seek placing specialized livestock in a rural cropping context, to avoid nutrient loading (in livestock producing areas) and nutrient depletion (in crop producing areas) that would occur otherwise. Better geographic distribution can be achieved by a variety of policy tools that can, and often need to, be combined. In developing countries, there will often be a need for investment in rural infrastructure (roads, electricity, slaughterhouses) to make rural areas attractive to large-scale livestock producers.

Zoning regulations and taxes can be used, for example, to discourage large concentrations of intensive production close to cities and far from cropland where nutrients could be recycled. In Thailand, high taxes were levied on poultry and pig production within a 100 kilometre radius of Bangkok, while areas further away enjoyed tax free status. This led to many new production units being established away from the major consumption centre. Improving the spatial distribution creates opportunities for waste recycling on land, which can simultaneously increase farm profits and reduce pollution (Gerber and Steinfeld, 2006). In the Netherlands, tradable manure quotas have been practiced until recently, so as to keep a ceiling on overall livestock density while providing a market mechanism to encourage efficiency.

Decision-support tools exist to assist policy-makers in designating zoning policies, taking into account environmental objectives and social and animal health considerations, while keeping in mind producers' requirements to operate profitably (Gerber *et al.*, 2006). This allows intensive production to be kept away from protected areas, human settlements, and surface water, and to be directed where there is arable land with a demand for nutrients, or where waste management is less of an environmental burden. Likewise, given that industrial livestock is a dynamic industry, which has become footloose with industrialization (Naylor *et al*, 2005) and moves where returns are most profitable. "Preferred zones" can, therefore, be designated so as to provide a growth stimulus to areas where this is lacking. Zoning is a particularly suitable instrument for the establishment of new operations, i.e. in areas with livestock sector growth; resettlement of already established farms has shown to be quite cumbersome. There is usually a need to combine zoning policies with licensing or certification schemes, so as to oblige operators to comply with environmental and other regulations before starting operations. Environmental licensing relies on nutrient management plans as an essential ingredient, which can be supported by appropriate models (for example LEAD, 2002).

Zoning is quite demanding in terms of institutional enforcement. It is usually combined with regulatory frameworks that include emissions standards for nutrients, biological oxygen demand, and pathogens; regulation of waste application (time, method, quantities); and regulations for feeding (use of antibiotics, copper, heavy metals, other feed quality). Regulations may vary by zone, and they may be more lenient where environmental problems are less pronounced. They may also be accompanied by training and extension programmes to acquaint farmers with the required knowledge and technologies.

A wide variety of management options exist to address pollution at various stages. Public policies need to encourage options that have been demonstrated to reduce nutrient loads and their environmental impact. These technical options were examined on Chapter 4 and include:

- manure separation and storage;
- lining of effluent ponds;
- provision of extra capacity to avoid overflows;
- optimizing land application of manure;
- close monitoring of nutrient flows;
- minimization of cleaning and cooling water;
- reduction of metal, antibiotic and hormone additives in feeds;
- optimal balancing of nutrients and improving feed conversion with enzymes and synthetic amino-acids; and
- biogas generation (which also reduces greenhouse gas emissions).

Such practices can be compiled into codes of conduct, as part of voluntary programmes, certification schemes or regulatory frameworks (see Box 6.4). Their application can also be facilitated through subsidy schemes, particularly for early adopters or when the adoption of these technologies requires investments, as is the case in many countries for biogas digesters. To capture the economies of scale in waste management, local authorities may encourage producers to form waste management groups and provide them with access to extension and training. Close monitoring of nutrient flows is crucial to nutrient management and enforcement of regulations.

The enforcement of environmental regulations to encourage or require adoption of advanced waste management technologies will affect production costs and competitiveness of farms to varying extents. Gerber (2006), modelling the costs of complying with environmental regulations for intensive livestock production in Thailand, found that profit reductions were limited (up to 5 percent) for farms with adequate access to land for waste application and advanced manure management technology. For those with no access to such land, profit reductions were higher, typically greater than 15 percent. This implies that differences in costs of compliance are likely to have an impact on where farms are located and, hence, on the geographical distribution of livestock.

6.2.4 Lessening the environmental impact of intensive feedcrop production

With 33 percent of all arable land dedicated to the production of feedcrops, livestock have an important environmental impact associated with

Box 6.4 Examples of successful management of livestock waste production from intensive agriculture

BELGIUM: LIVESTOCK WASTE MANAGEMENT STARTS AT THE FRONT AND NOT AT THE BACK OF THE ANIMAL

The government of the Flemish part of Belgium introduced a three-track strategy to reduce the excess of 36 million kg phosphate and 66 million kg nitrogen discharged in its soil and water. It consisted of (a) reducing livestock numbers and reducing nutrient intake by providing low-protein and phosphate feeds. The latter was introduced on the basis of a voluntary agreement between the government and the feed miller association, (b) manure processing and export, and (c) improving manure management. It was expected that the first two would reduce the phosphate surplus each by 25 percent, and that improved manure management by half. However, by 2003, when the P_2O_5 surplus was reduced to 6 million kg, measure (a) had contributed with 21 million kg (of which 13 million kg from improved feed technology, whereas (b) and (c) together had contributed only 7.5 million kg. The total reduction of 41 million kg, of nitrogen, 11 million was the result of low protein diets, demonstrating the potential optimal rationing of N and P in reducing nutrient loading.

Source: Mestbank (2004).

THE NETHERLANDS: LINKING ENVIRONMENT AND COMMERCE – INTRODUCING A MANURE QUOTA SYSTEM

A system of manure production quotas was established in the Netherlands in 1986. The quota was based on historical standard manure production amounts per animal. Farmers were allocated a manure production quota, expressed in kg P_2O_5. The manure production rights were made tradable in 1994, and supported by a mineral accounting system, and strict regulations on application techniques. Despite its significant administrative burden, and high cost to intensive livestock farms, the results are impressive, as the loading of the soil with N and P decreased substantially over time. Reduced application of mineral fertilizer also contributed to that. Between 1998 and 2002, the net loading of the soil decreased by 169 million kg per year for N and by 18 million kg per year for P. The net loading of the soil decreased by about 0.2 kg P and 0.8 kg N per euro spent (RIVM, 2004). The cost of removal of N and P from surface waters are much higher.

Source World Bank (2005).

intensive agriculture, and with the expansion of arable land into areas not previously cropped, in particular forests. The large-scale production of crops for feed is currently concentrated mostly in Europe, North America, parts of Latin America and Oceania. Expansion of cropland for feed is strongest in Brazil, in particular for soybeans, but it is also occurring in many developing countries, mostly in Asia and Latin America. The bulk of global feedcrops is produced under commercial and mechanized conditions. Smallholders play only a local role in supplying grains and other crops for feed.

The key to reducing the pollution and other environmental impacts associated with intensive agriculture for feed production lies in increasing efficiency that is, increasing production while reducing inputs that have environmental impacts, including fertilizer, pesticides and fossil fuel. Advanced technology has shown remarkable progress in some areas. For example, fertilizer and pesticide use has declined substantially in many developed countries at the same time as yields have continued to grow.

Research and regulatory frameworks have been instrumental in bringing down fertilizer application rates and in limiting pollution from fertilizer in most developing countries, by developing and disseminating slow release and other less polluting formulations, tightening emission and discharge standards for fertilizer factories, higher fines, placing physical limits on the use of manure and mineral fertilizers and by application of the nutrient budget approach (FAO, 2003). Since the early 1990s developed countries have also started to introduce economic measures in the form of pollution taxes on mineral fertilizers. A number of developing countries still subsidize mineral fertilizer production or sales, either directly or indirectly (as energy subsidies to nitrogen fertilizer producers). The use of low-efficiency fertilizers such as ammonium carbonate needs to be discouraged.

Pesticide use is rapidly increasing in many emerging economies, whereas it is declining from high levels in most developed countries. Policies to address excessive pesticide use include testing and licensing procedures for pesticides before they are allowed on the market (FAO, 2003). Environmental problems that arise from the accumulation of pesticide residues in soils and in water need to be monitored, preferably by independent institutions. The imposition of pollution taxes on pesticides creates economic incentives to reduce their use.

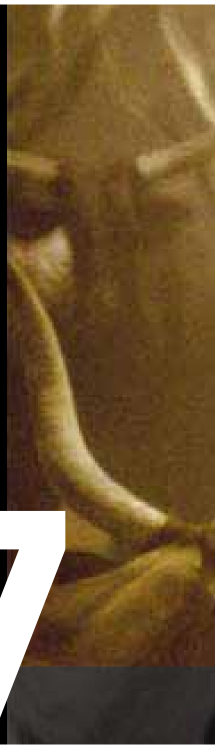
For areas that are experiencing expansion of arable land for feed production, into areas not previously cropped, there is a need to facilitate the land-use transition. The most suitable and productive areas need to be intensified and marginal areas retired into stable pastures or forest land. This process can be assisted by land titling and zoning policies, by targeted research and extension work, and by selected infrastructure development.

Targeted research and extension can also help in promoting more environmentally benign cultivation methods, including conservation agriculture or no-tillage systems and forms of organic farming. Precision agriculture, which uses advanced information and satellite technology to tailor the amount and timing of inputs to specific small areas, has been shown to have substantial potential for further productivity increases, while limiting and optimizing input use.

Since a large part of the feed-producing area is irrigated, particularly for dairy production where there is a need for fresh fodder, water is an important input that is greatly affected by livestock feed demand. Pricing, establishing water markets and building appropriate institutional frameworks, as discussed previously, are indispensable policy instruments for achieving higher water use efficiencies and for addressing depletion.

A different pathway to addressing the environmental impact of feedcrop production is to reduce demand. As has already been discussed in earlier chapters, this can be achieved by creating policy conditions to promote the use of advanced technologies to improve feed efficiency, such as phased feeding, the use of enzymes such as phytase and phosphatase, use of synthetic amino-acids and other feed ingredients. These inputs are sometimes subject to tariffs. A reduction, or elimination, of such trade barriers may facilitate the uptake of related technologies.

07



Summary and conclusions

As we have seen, the livestock sector is a major stressor on many ecosystems and on the planet as whole. Globally it is one of the largest sources of greenhouse gases and one of the leading causal factors in the loss of biodiversity, while in developed and emerging countries it is perhaps the leading source of water pollution.

The livestock sector is also a primary player in the agricultural economy, a major provider of livelihoods for the poor and a major determinant of human diet and health. Hence its environmental role needs to be seen in the context of its many different functions, in many diverse natural and economic environments, subject to diverse policy objectives.

Previous chapters have described the state of knowledge about livestock–environment interactions at local, regional and global scales. This chapter puts forward possible future scenarios for the sector. What are societies' expectations of the livestock sector? What are the differences between countries and how are these expectations changing over time?

The necessary steps towards shrinking livestock's long shadow are outlined. Mastering the political will to implement these steps obviously hinges on the question: what relative value should we assign to the environment, compared to other objectives such as the provision of livelihoods or the cheap supply of animal products?

And, if we do rate environmental considerations as important, how can public attention be moved beyond the more obvious, but less serious "nuisance" of flies and odour, to the more important pressures of land degradation, water pollution, biodiversity erosion and global climate change?

7.1 Livestock and environment in context

Chapter 6 presented the conflicting policy objectives schematically. Policy decisions will be based largely on the economic, social, health and food security considerations as summarized below.

Economic importance

Heading for over half of agricultural GDP

As an economic activity, the livestock sector generates about 1.4 percent of the world's GDP (2005). The sector's growth rate of 2.2 percent for the last ten years (1995 to 2005) is roughly in line with overall economic growth (FAO, 2006b). It is growing faster than the GDP of agriculture, which is declining in terms relative to overall GDP. Currently, the livestock sector's GDP accounts for a global average of 40 percent of agricultural GDP, and shows a strong tendency to increase towards the 50 to 60 percent range that is typical for most industrialized countries. The livestock sector provides primary inputs (raw milk, live animals, etc.) to the agricultural and food industry, where value-adding activities multiply the value of these raw materials.

Social importance

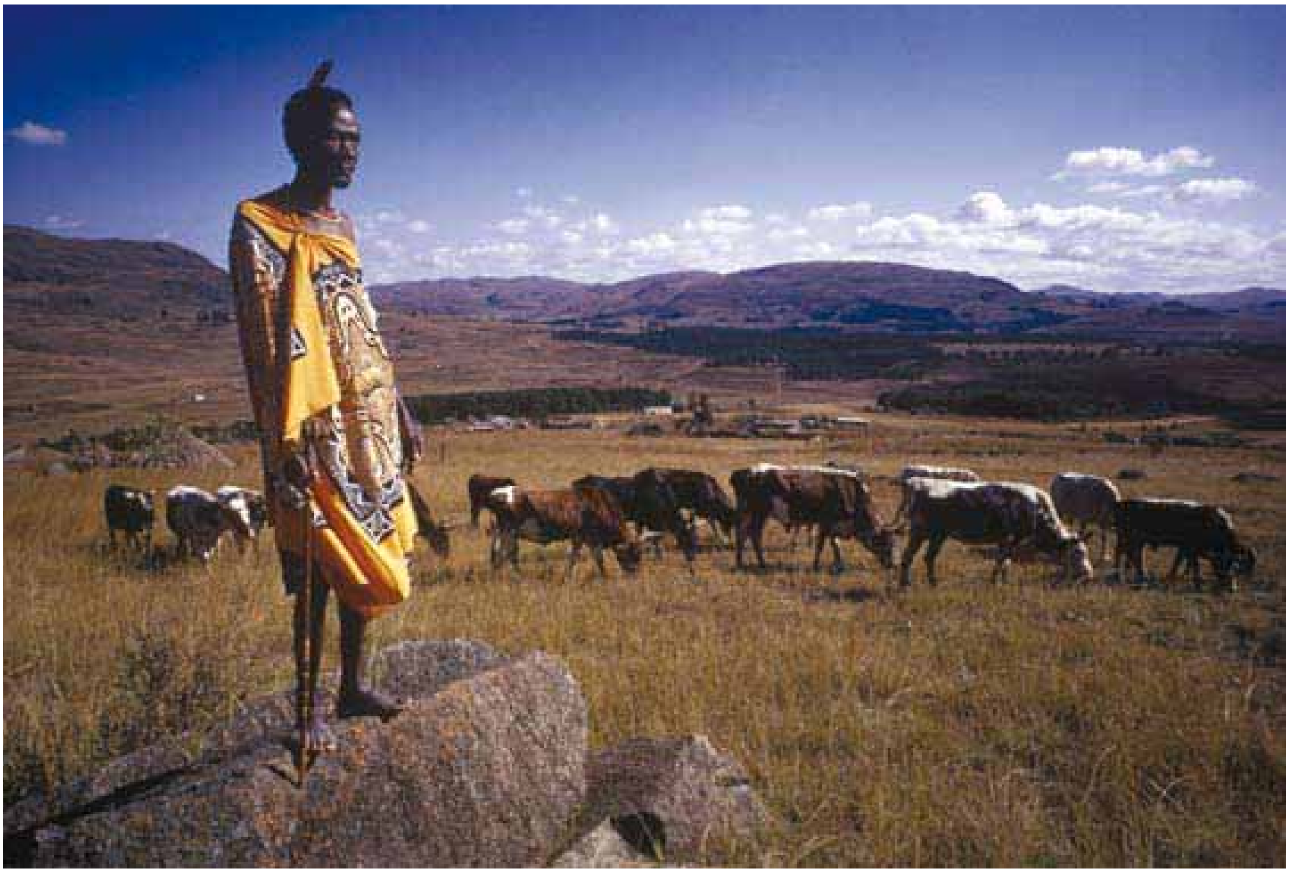
Livelihoods for one billion poor

In terms of livelihood support, income and employment, the livestock sector is much more important than its modest contribution to the overall economy would suggest. Livestock provide livelihood support to an estimated 987 million poor people in rural areas (Livestock In Development, 1999), equivalent to 36 percent of the total number of poor, currently estimated at 2 735 million (i.e. people living on less than US$2 per day) (World Bank, 2006). As livestock rearing does not require formal education or large amounts of capital, and often no land ownership, it is often the only economic activity accessible to poor people in developing countries. In many marginal areas of developing countries, livestock production is an expression of the poverty of people who have no other options, and do not have the means to counteract environmental degradation either. The huge number of people involved in livestock for lack of an alternative, particularly in Africa and Asia, is a major consideration for policy-makers, and any attempts to address livestock-associated environmental degradation must take these livelihood concerns into account. In contrast, in the developed countries decades of continuous structural change have reduced the number of people engaged in livestock production, which is more in line with the sector's modest economic contribution.

Decision-making in the livestock sector is often complicated by the important socio-cultural roles that livestock continue to play in many societies. These take different forms and include livestock as an expression of wealth and prestige, as a method of payment (bride price and dispute settlement) and risk diversion for mixed crop-livestock farmers, etc. Food preferences and taboos relate in a particular way to products of animal origin.

© UNDP / 6537 / T. FINCHER

Milk offers a good way of providing a protein-rich diet for the mass of Indian people, a great number of whom are vegetarians – India 1977



A headman looks over his cattle – Swaziland 1971

Nutrition and health

A major determinant

In terms of nutrition, livestock food products globally contributed an average of 17 percent of energy and 33 percent of protein to dietary intakes in 2003 (FAO, 2006b). There are stark differences between countries and country groups, with meat consumption ranging in 2003 from only 5 kg per person and year in India to 123 kg in the United States (FAO, 2006b). Because developing countries still have low intakes of animal food the share of livestock products in the "global average diet" is expected to continue to rise to reach the OECD country averages of about 30 percent of dietary energy and 50 percent of protein intake. In terms of health and nutrition, therefore, livestock products are a welcome addition to the diets of many poor and under- or malnourished people who frequently suffer from protein and vitamin deficiencies as well as from lack of important trace minerals. Children in particular have shown to benefit greatly in terms of physical and mental health when modest amounts of milk, meat or eggs are added to their diets, as shown by long-term research carried out in Kenya (Neumann, 2003). In contrast, a large number of non-communicable diseases among the more wealthy segments of the world's population are associated with high intakes of animal source foods, in particular animal fats and red meat: cardio-vascular disease, diabetes and certain types of cancer. While not being addressed by this assessment, it may well be argued that environmental damage by livestock may be significantly reduced by lowering excessive consumption of livestock products among wealthy people. International and national public institutions (e.g. WHO and Tufts University, 1998) have consistently recommended lower intakes of animal fat and red meat in most developed countries.

In terms of health and food safety, livestock products as a category are more susceptible to pathogens than other food products. They have the capacity to transmit diseases from animals to humans (zoonoses). The World Organization for Animal Health (OIE) estimates that no less than 60 percent of human pathogens and 75 percent of recent emerging diseases are zoonotic. A series of human diseases have their known origins in animals (such as common influenza, small pox). Tuberculosis, brucellosis and many internal parasitic diseases, such as those caused by tapeworm, threadworm and so on, are transmitted through the consumption of animal products. Recent emerging diseases, such as avian flu, Nipah virus or the variant Creutzfeldt-Jakob disease demonstrate the potential of the human- livestock interface to develop and transmit novel diseases. Therefore, sanitary concerns are of paramount importance in the livestock industry, particularly when the requirements of long and sophisticated food chains govern the retail sector as is the case in OECD countries and increasingly in developing countries. Human and animal health concerns are a major driving force for structural change in the livestock sector. In the case of animal health, control of major disease is greatly facilitated by, and sometimes impossible without, confinement of animals and animal movement control.

Food security

Livestock compete for crops but provide a buffer against grain shortages. In simple numeric terms, livestock actually detract more from total food supply than they provide. Livestock now consume more human edible protein than they produce. In fact, livestock consume 77 million tonnes of protein contained in feedstuff that could potentially be used for human nutrition, whereas only 58 million tonnes of protein are contained in food products that livestock supply. In terms of dietary energy, the relative loss is much higher. This is a result of the recent trend towards more concentrate-based diets for pigs and poultry, with nutritional requirements more similar to humans than ruminants.

This simple comparison obscures the fact that proteins contained in animal products have higher nutritive values than those in the feed provided to animals. Moreover, it does not capture the fact that livestock and their feed also make a contribution to food security objectives by providing a buffer in national and international food supplies that can be drawn upon in case of food shortages. However, as the livestock sector moves away from using feed and other resources that have no or little alternative value, towards using crops and other high value inputs, it enters into competition with food and other uses of commodities and land. While it is probably true that livestock do not detract food from those who currently go hungry, it raises overall demand and prices for crops and agricultural inputs.

These various aspects of livestock's importance feed into national decision-making for the sector. The different policy objectives of food supply, poverty reduction, food safety and environmental sustainability take on different levels of importance depending on factors such as stage of development, per capita income and general policy orientation of a country. In least developed countries with large smallholder sectors, concerns of small producers weigh heavily, along with those of providing cheap supplies to urban consumers. In higher income countries, consumer concerns for food and environmental safety usually override producer interests, even though governments continue to support and protect domestic production for a variety of reasons (see Chapter 6).

There is a stark contrast between the rather modest economic contribution of the livestock sector and its important social, environmental and health dimensions. It is against this background that livestock-environment interactions need be seen. These are the facts that emerge:

Land and land-use change

Humanity's largest land use

Livestock's land use includes grazing land and cropland dedicated to the production of feedcrops and fodder. In fact livestock represent the largest of all anthropogenic land uses. The total

Table 7.1

Global facts about livestock

Dimension	Parameter	Value	Remarks
Economic importance[a]	Contribution to total GDP (2005)	1.4 percent	
	Contribution to agricultural GDP (2005)	40 percent	
	Growth rate (1995 to 2005)	2.2 percent p.a.	
	Contribution to agricultural export earnings (2004)	17 percent	
Social importance[b]	Number of poor engaged in livestock activities	987 million	Full time or partially
	Total number of people engaged in livestock production	1 300 million or 20 percent of world population of 6.5 billion	Full time or partially
Food security[c]	Human edible protein supplied to livestock[1]	77 million tonnes	
	Human edible protein supplied by livestock[1]	58 million tonnes	
Health[c]	Contribution to total dietary intake of energy[d]	477 kcal per person/day or 17 percent of average daily intake	
	Contribution to total dietary intake of protein[d]	25 g per person/day or 33 percent of average daily intake	
	People suffering from under or malnourishment[2]	864 million	Livestock products are a possible remedy
	Number of overweight persons[3]	1 000 million	Livestock products are one of the major causes
	People suffering from obesity[3]	300 million	Livestock products are one of the major causes
Environment: land[e]	Total land for grazing	3 433 million ha or 26 percent of terrestrial surface	
	Grazing land considered degraded	20 to 70 percent	
	Total land for feed crop cultivation[4]	471 million ha or 33 percent of arable land	
Environment: air and climate[5]	Livestock's contribution to climate change in CO_2 equivalent	18 percent	Incl. pasture degradation and land use change
	Livestock's share in carbon dioxide emissions	9 percent	Not considering respiration
	Livestock's share in methane emissions	37 percent	
	Livestock's share in nitrous oxide emissions	65 percent	Including feed crops
Water[6]	Share of livestock in total use of freshwater	8 percent	Drinking, servicing, processing and irrigation of feed crops
	Share of livestock in water evapotranspirated in agriculture	15 percent	Evapotranspiration for feedcrops production only; other factors siginifcant but not quantifiable

[1] Protein content derived by applying the appropriate protein nutritive factors to respective input and output commodities.
[2] Three-year average 2002-04.
[3] Data refers to adult population.
[4] See Chapter 2 and Annex 3.1.
[5] See Chapter 3.
[6] See Chapter 4.

Sources: [a] World Bank (2006) and FAO (2006b); [b] Livestock In Development (1999); [c] FAO (2006b); [d] Data on livestock contribution to protein and energy dietary intake: FAO (2006b); data on malnourishment: Food Security – FAO (2006b); data on obesity and overweight: World Health Organization, 2006. [e] FAO (2006b).

areas involved are vast, amounting to 70 percent of all agricultural land and 30 percent of the ice-free terrestrial surface of the planet.

The total land area occupied by livestock grazing is 3 433 million hectares equivalent to 26 percent of the ice-free terrestrial surface of the planet. A large part of these areas are too dry or too cold for crop use, and only sparsely inhabited. While the total grazing area is not increasing, in tropical Latin America there is rapid expansion of pastures into some of the most vulnerable and valuable ecosystems, with 0.3 to 0.4 percent of forest lost to pastures annually. In the Amazon, cattle ranching is now the primary reason for deforestation. In contrast, in developed countries, forest areas are growing as marginal pastures are afforested, but the biodiversity and climate change value of these forest areas gained in developed countries are much inferior to those lost in tropical areas.

About 20 percent of the world's pastures and rangeland have been degraded to some extent, but 73 percent of rangeland in the dry areas (UNEP, 2004b). The Millennium Ecosystem Assessment has estimated that 10 to 20 percent of all grassland is degraded. Some of the dryland grazing ecosystems have proved to be quite resilient and degradation has shown to be reversible in parts.

The total area dedicated to feedcrop production amounts to 471 million hectares, equivalent to 33 percent of the total arable land. Most of this total is located in OECD countries, but some developing countries are rapidly expanding their feedcrop production, notably maize and soybean in South America, in particular in Brazil. A considerable part of this expansion is taking place at the expense of tropical forests. It is expected that future growth rates of livestock output will be based on similar growth rates for feed concentrate use (FAO, 2006a). Intensive feed production is often associated with various forms of land degradation, including soil erosion and water pollution.

Gaseous emissions and climate change

More impact than road transport

Here too livestock's contribution is enormous. It currently amounts to about 18 percent of the global warming effect – an even larger contribution than the transportation sector worldwide. Livestock contribute about 9 percent of total carbon dioxide emissions, but 37 percent of methane and 65 percent of nitrous oxide.

Greenhouse gases are emitted from rumen fermentation and livestock waste. Carbon dioxide is released when previously forested areas are converted into grazing land or arable land for feed. Therefore, expansion of pasture and cropland at the expense of forests releases significant amounts of carbon dioxide into the atmosphere. As does the process of pasture and arable land degradation, which results in a net loss of organic matter. Carbon dioxide releases resulting from fossil fuel consumption used for the production of feed grains (tractors, fertilizer production, drying, milling and transporting) and feed oil crops must also be attributed to livestock. The same applies with the processing and transport of animal products. Yet another category is constituted by nitrous oxide emissions from leguminous feedcrops and from chemical fertilizer applied to other feedcrops.

In terms of polluting gaseous emissions not linked to climate change, livestock waste emits a total of 30 million tonnes of ammonia. This is focused in areas of high animal concentrations, where ammonia is a factor in the occurrence of acid rain, which affects biodiversity. Livestock contribute 68 percent to total ammonia emissions.

Water

A major driver of use and pollution

The livestock sector is a key player in increasing water use and water depletion. The water used by the livestock sector is over 8 percent of global human water use. The major part of this water is in fact used for irrigation of feed crops, representing 7 percent of the global water use. The water used for product processing and drinking and servicing is insignificant at global level (less than 1 percent of the global water), but it may be of local importance in dry areas (livestock drinking requirements represent 23 percent of total water use in Botswana).

Apart from livestock's use of water for drinking, water is used for irrigating pastures and cropland for feed production. Considerable amounts of water are used in processing of meat and milk in particular. Through the compacting effect of grazing and hoof action on the soil, livestock also have a determining, and often negative, impact on water infiltration and the speed of water movement across the landscape. Livestock play an important role in water quality through the release of nutrients, pathogens and other substances into waterways, mainly from intensive livestock operations.

The contribution of the livestock sector to water depletion is not easily quantified with our current knowledge but there is strong evidence that the sector is a major driver. The volume of water evapotranspired by feedcrops represents a significant share (at 15 percent) of the water depleted every year.

Water pollution figures from the United States, the world's largest economy and fourth largest land area, may give some indication of the livestock sector's importance. In the United States, livestock are responsible for an estimated 55 percent of erosion, 37 percent of the pesticides applied, 50 percent of the volume of antibiotics consumed and for 32 percent of the nitrogen load and 33 percent of the phosphorus load into freshwater resources. Although the effective load into freshwater resources is not assessed for sediments, pesticides, antibiotics, heavy metals or biological contaminants, livestock are likely to have a major role in these pollution processes.

Livestock land use and management (especially of animal wastes) appear to be the main mechanism through which livestock contribute to the water depletion process.

Biodiversity

Livestock are a key factor in loss of species

Livestock affect biodiversity in many direct and indirect ways, most of which are difficult to quantify. Livestock and wildlife interact in grazing areas, often negatively, sometimes positively. Livestock help to maintain some of the open grassland ecosystems in their traditional state, but health concerns pose new threats to wildlife.

Pasture expansion, often at the expense of forest, has vast negative consequences on some of the most valuable ecosystems in Latin America, while rangeland degradation affects biodiversity on all continents. Crop area expansion and intensification for livestock feed undoubtedly affect biodiversity negatively, sometimes with dramatic consequences (soybean expansion into tropical forests). Water pollution and ammonia emissions, mainly from industrial livestock production, compromise biodiversity, often drastically in the case of aquatic life. Livestock's important contribution to climate change will clearly have repercussions on biodiversity, while the historic role of livestock as a driver and facilitator of invasions by alien species continues.

Livestock now account for about 20 percent of the total terrestrial animal biomass, and occupies a vast area that was once habitat for wildlife. Further, livestock determine, to a significant extent, the nitrogen and phosphorus flows. The fact that the livestock sector is industrializing, in a number of concentrated locations, separates the sector from its land base and interrupts the nutrient flows between land and livestock, creating problems of depletion at the sources (land vegetation and soil) and problems of pollution at the sinks (animal wastes, increasingly disposed of into waterways instead of back on the land). Pollution, as well as overfishing for feed, leads to an increasingly strong impact of livestock on the biodiversity of marine ecosystems.

Differences between species, products and production systems

There are huge differences in environmental impact between the different forms of livestock production, and even the species.

Cattle provide a multitude of products and services, including beef, milk, and traction. In mixed farming systems, cattle are usually well integrated in nutrient flows and can have a positive environmental impact. In developing countries, cattle and buffaloes still provide animal draught for field operations, and in some areas, animal traction is on the increase (parts of sub-Saharan Africa) so that animals substitute for potential fossil fuel use. Livestock also use crop-residues some of which would otherwise be burned, thus making net contributions to environmental objectives. However, cattle in extensive livestock production in developing countries are often only of marginal productivity. As a result, the vast majority of feed is spent on the animal's maintenance, leading to resource inefficiencies and high levels of environmental damage per unit of output.

The dairy sector is much better connected to land than is the case for other forms of market-oriented production. Most milk operations tend to be close to areas of feed supply because of their daily demand for fibrous feed, and so they are predominantly well integrated with nutrient flows, although excessive use of nitrogen fertilizer on dairy farms is one of the main causes of high nitrate levels in surface water in OECD countries. There is a risk of soil and water contamination by large-scale dairy operations, as witnessed by "dairy colonies" in South Asia, and by industrial-type operations in North America and increasingly also in China. Dairy production is also labour-intensive and less subject to economies of scale. Therefore, dairy is the livestock commodity where small-scale or family-based operations can resist market pressures for longer than is the case for poultry or pork.

Beef is produced in a wide range of intensities and scales. At both ends of the intensity spectrum there is considerable environmental damage. On the extensive side, cattle are instrumental in degradation of vast grassland areas and are a contributing factor to deforestation (pasture conversion), and the resulting carbon emissions, biodiversity losses and negative impacts on water flows and quality. On the intensive side, feedlots are often vastly beyond the capacity of surrounding land to absorb nutrients. While in the feedlot stage the conversion of concentrate feed into beef is far less efficient than into poultry or pork, and therefore beef has significantly higher resource requirements per unit than pork or poultry. However, taking the total life cycle into account, including the grazing phase, concentrate feed per kilogram of growth is lower for beef than for non-ruminant systems (CAST, 1999).

The production of sheep and goats is usually extensive. Except for small pockets with feed lots in Near East and North America, intensive production based on feed concentrate barely exists. The capacity of small ruminants, in particular goats – to grow and reproduce under conditions otherwise unsuitable for any form of agricultural production – makes them useful and very often essential to poor farmers pushed into these environments for lack of alternative livelihoods. Because of their adaptive grazing, sheep and goats have extended their reach further into arid, steep and otherwise marginal territory than cattle. The browsing of goats affects land cover and the potential for forest re-growth. Under overstocked conditions, they are particularly damaging to the environment, through degradation of vegetative cover and soil. However, the low economic value of sheep and goat production means that it does not usually lead directly to mechanized large scale deforestation, as is the case for cattle ranching in Brazil.

Extensive pig production, based on use of household waste and agro-industrial by-products, performs a number of useful environmental functions by turning biomass of no commercial value – and that otherwise would be waste - into high-value animal protein. However, extensive systems are incapable of meeting the surging urban demand in many developing countries, not only in terms of volume but also in sanitary and other quality standards. The ensuing shift towards larger-scale grain-based industrial systems has been associated with geographic concentration, to such extents that land/livestock balances have become very unfavourable, leading to nutrient overload of soils and water pollution. China is a prime example of these trends. Furthermore, most industrial pig production in the tropics and sub-tropics uses waste-flushing systems involving large amounts of water. This becomes the main polluting agent, exacerbating negative environmental impact.

Poultry production has been the species most subject to structural change. In OECD countries, production is almost entirely industrial, while in developing countries it is already predominantly industrial. Although industrial poultry production is entirely based on feed grains and other high value feed material, it is the most efficient form of production of food of animal origin (with the exception of some forms of aquaculture), and has the lowest land requirements per unit of output. Poultry manure is of high nutrient content, relatively easy to manage and widely used as fertilizer and sometimes as feed. Other than for feedcrop production, the environmental damage, though perhaps locally important, is of a much lower scale than for the other species.

In conclusion, livestock-environment interactions are often diffuse and indirect; and damage occurs at both the high and low end of the intensity spectrum, but is probably highest for beef and lowest for poultry.

7.2 What needs to be done?

The future of the livestock-environment interface will be shaped by how we resolve the balance of two competing demands: for animal food products on the one hand and for environmental services on the other. Both demands are driven by the same factors: increasing populations and increasing incomes and urbanization. The natural resource base within which they must be accommodated is finite. Therefore, the considerable expansion of the livestock sector required by expanding demand must be accomplished while substantially reducing livestock's environmental impact. In this section we put forward perspectives as to how this can be achieved, compared to a backdrop of "business as usual."

The growth in demand for animal products over the coming decades will be significant. Although the annual growth rate will be somewhat slower than in recent decades, the growth in absolute volume will be vast. Global production of meat is projected to more than double from 229 million tonnes in 1999/2001 to 465 million tonnes in 2050, and that of milk to increase from 580 to 1 043 million tonnes (FAO, 2006a). The bulk of the growth in meat and in milk production will occur in developing countries (FAO, 2006a). Among the meat products, poultry will be the commodity of choice for reasons of acceptance across cultures and technical efficiency in relation to feed concentrates.

Business as usual leads to mounting problems

In the absence of major corrective measures, the environmental impact of livestock production will worsen dramatically. Viewed very simply, if production doubles, without any reduction in environmental measures per unit of production, then environmental damage will double.

Taking into account likely changes in the structure of the industry, while there has been no attempt to quantify the environmental impacts of livestock, it is probably safe to state that under a "business as usual" scenario:

- The spatial and commercial concentration of livestock production will continue to grow, leading to large areas with high nitrogen and phosphorus surpluses, concentrated discharge of toxic materials, polluting and contaminating land and ground and surface water, and destroying terrestrial and aquatic biodiversity. Continued geographic concentration, with large-scale commercial production growing but with less intensive, widely scattered smallholder production still existing alongside, will exacerbate the risk of emerging and traditional zoonotic diseases.

- Demand for feedcrops will grow, causing a further conversion of natural habitats into cropland in some places, notably Latin America. The factors that slowed use of feedgrain in the period 1985 to 2005, including EU agricultural policy reform, drastic structural changes in the previous socialist countries of Eastern Europe and CIS, and the global shift to poultry as efficient converters of feed crops, are likely to wane (FAO, 2006a); therefore feedgrain use is projected to expand more in line with output growth in livestock products. The pressure on crop agriculture to expand and intensify will remain high; and so the associated environmental impacts, in terms of water depletion, climate change and biodiversity loss, will grow.
- Livestock's contribution to anthropogenic greenhouse gas emissions will increase, in particular of the more aggressive nitrous oxide, raising the sector's already significant contribution to global climate change; and
- Livestock-induced degradation of the world's arid and semi-arid lands will continue, in particular in Africa and South and Central Asia, again contributing significantly to climate change, water depletion and biodiversity losses, and sometimes leading to irreversible loss of productivity. The poor who derive a living from livestock will continue to extract the little they can from dwindling common property resources while facing growing marginalization.

Consumers may drive change towards a sustainable livestock sector

These "business as usual" trends lead to disaster and need to be diverted into more beneficial paths. Growing economies and populations combined with increasing scarcity of environmental resources and rising environmental problems are already translating into a growing demand for environmental services. Increasingly, this demand will broaden from immediate factors of concern, such as reducing the nuisance factors of flies and odours, to the intermediate demands of clean air and water, to the broader, longer-term environmental concerns, including climate change, biodiversity, etc. At the local level, markets will undoubtedly develop for the provision of such services; this is already the case for water in many places. At the global level, this is more uncertain although promising models already exist, for example carbon trading or debt-for-nature swaps.

There are reasons for optimism that the conflicting demands for animal products and environmental services can be reconciled. Both demands are exerted by the same group of people, the relatively affluent, middle to high level income class, which is no longer confined to industrialized countries. It has already firmly established itself in a number of developing countries, and is poised to grow substantially in most developing countries over the coming decades. This group of consumers is probably ready to use its growing voice to exert pressure for change and may be willing to absorb the inevitable price increases. The development of markets for organic products and other forms of eco-labelling are precursors of this trend, as are the tendency towards vegetarianism within developed countries and the trend towards healthier diets.

Encouraging efficiency through adequate market prices

Resource-use efficiency is the key to shrinking livestock's long shadow. A host of tested and successful technical options are available to mitigate environmental impacts, which can be used in resource management, in crop and livestock production, and in post harvest reduction of losses. They have been summarized in the various chapters of this assessment. However, for them to be widely adopted and applied will require adequate price signals, more closely reflecting the true scarcities of production factors, and correcting the distortions that currently provide insufficient incentives for efficient resource use.

Prices of land, water and feed resources used for livestock production do not reflect true scarcities. This leads to an overuse of these resources by the livestock sector and to major inefficiencies in the production process. Any future policy to protect the environment will, therefore, have to introduce adequate market pricing for the main inputs.

In particular, water is grossly under-priced in most countries. The development of water markets and different types of cost recovery have been identified as suitable mechanisms to correct the situation. In the case of land, suggested instruments include the introduction and adjustment of grazing fees and lease rates, and improved institutional arrangements for controlled and equitable access. Further, the removal of price support at product level (i.e. the production subsidies for livestock products in the majority of industrialized countries) is likely to improve technical efficiency. This is shown, for example, in New Zealand where in the early eighties radically cut agricultural subsidies, resulting in what has become one of the most efficient and environmentally benign ruminant livestock industries.

Correcting for environmental externalities

Although the removal of price distortions at input and product level will go a long way to enhancing the technical efficiency of natural resource use in the livestock production process, this may often not be sufficient. Environmental externalities, both negative and positive, need to be explicitly factored into the policy framework, through the application of the "provider gets - polluter pays" principle.

Correcting for externalities, both positive and negative, will lead livestock producers into management choices that are less costly to the environment. Livestock holders who provide environmental services need to be compensated, either by the immediate beneficiary (such as with improved water quantity and quality for downstream users) or by the general public. Examples of actions that could be rewarded include land management and use forms and vegetative covers that maintain or restore biodiversity; or the sequestration of carbon in stable organic matter in the soil through pasture management. Managing grasslands in order to reduce runoff and increase infiltration can greatly reduce sedimentation of water reservoirs: compensation schemes need to be developed between water and electricity providers and grazers.

Likewise, livestock holders who emit waste into waterways or release ammonia into the environment must be held accountable and pay for the damage, to encourage them to move to less polluting practices. Applying the polluter pays principle should not present insurmountable problems in situations like these, given that burgeoning demand for livestock products provides the potential for adequate profits, and that there is an increasing demand for milk and meat produced in a sustainable way. It will be difficult to apply this principle to methane emissions from single cows owned on an Indian mixed farm of half a hectare. However, for most waste emissions in intensive production units, a combination of disincentives and regulation seems to be the most appropriate approach.

It is expected that the taxation of environmental damage and incentives for environmental benefits will be much more rigorously applied in future, tackling local externalities first but increasingly also trans-boundary impacts, through the application of international treaties, underlying regulatory frameworks and market mechanisms. Government policies may be required to provide incentives for institutional innovation in this regard.

Accelerating technological change

In industrial, and mixed production systems, the gap between current levels of productivity and levels that are technically attainable indicates that important efficiency gains can be realized by adopting intensifying technologies. With extensive grazing, this is more difficult, sometimes even impossible – particularly under marginal conditions with severe resource constraints (such as in the Sahel), where current low productivity may be the maximum that can be achieved (Breman and de Wit, 1983). Intensification would be possible only on a limited area, estimated at about 10 percent of the total pasture area (Pretty *et al.*, 2000).

Correcting for distortions and externalities will bring us a step closer to prices for both inputs and outputs that reflect the true scarcities of production factors and natural resources used. These changed prices will induce technological change that will make better use of resources, and limit pollution and waste. Producers have shown their ability to respond quickly and decisively when such price signals are sent consistently.

For now there does not appear to be a problem of lack of improved production technologies. Given the large market, and policy failures, under which the livestock sector operates, there is still a huge amount of progress that can be achieved from wide adoption of existing tried and tested technologies. However, there is a continuing need for research and development of new technologies suited to more conducive policy frameworks.

Technological change needs to be driven towards making optimal use of land and water as the most important production factors for livestock, including feed production. Research and development for feed crop production need to further increase yields and factor efficiency. However, this is beyond the scope of this study.

In the livestock sector, the quest for increasing efficiencies mainly falls on feeding, breeding and animal health. The application of modern feeding techniques, in production systems that are already industrial but technologically not very advanced, can help reduce feed grain consumption significantly – perhaps by as much as 120 million tonnes, or 20 percent of total feed grain use (assuming that half of the yield gap between top feed performers and world averages can be closed). Such improvements would include the use of optimized rations, enzymes and artificial amino acids. Further savings in the grain bill could come from the use of advanced animal genotypes. While research into technological advances for commercial and industrial livestock production have been largely left to the private sector, the public sector needs to assume a proactive role in research and technology development with regard to natural resource management, and in reducing market barriers for small producers.

Reducing the environmental and social impacts of intensive production

As described in Chapter 1, an estimated 80 percent of total livestock sector growth comes from industrial production systems. The environmental problems created by industrial systems do not derive from their large scale or their production intensity, but from their geographical location and concentration. In extreme cases, size may be a problem: sometimes units are so large (a few hundred thousand pigs, for example) that waste disposal will always be a problem, no matter where these units are put.

Industrial systems are often located in a way that prevents sustainable waste management. Crop production and livestock activities are being increasingly separated, so that sufficient land to safely dispose of waste is not available nearby. So far, environmental concerns have not often been a factor shaping the regional distribution of livestock production. Easy access to input and product markets, and relative costs of land and labour have so far been the major determining factors. For developing countries, the concentration of industrial units in peri-urban environments is typical because of infrastructure constraints. In developed countries, there is certainly a move towards rural environments but this often seems to be motivated by an attempt to hide these places away, rather than addressing the fundamental environmental concerns. However, limitations on livestock densities (as introduced by the EU) have been a strong factor in arriving at a better balance between livestock and the surrounding ecosystem.

What is required therefore is to bring waste generated into line with capacity of accessible land to absorb that waste. Industrial livestock must be located as much as possible where cropland within economic reach can be used to dispose of the waste, without creating problems of nutrient loading – rather than geographically concentrating production units in areas favoured by market access, or feed availability. Suitable policy options include zoning and licensing, mandatory nutrient management plans, and facilitation of contractual agreements between livestock producers and crop farmers.

Only a spatially decentralized livestock sector will create sufficient opportunities and incentives for recycling livestock waste on land. For the medium-term future, the preferred option is the reintegration of crop and livestock activities. Policies need to drive the decentralization of industrial and intensive livestock away from consumption centres and ports, towards rural areas with nutrient demand. Such policies must comprise regulatory and incentive frameworks. Regulations are needed to deal with heavy metal and drug residue issues at the feed and waste levels, and with other public health aspects such as food-borne pathogens.

Spatially decentralized livestock activities can also offer substantial social benefits for rural development, particularly in areas with limited alternative employment and growth opportunities. Incentives need to accompany these regulations, such as lower taxes for establishment of commercial production units in nutrient deficit areas, eventually subsidies for relocation of large scale enterprises.

Where decentralization cannot be achieved, industrial systems need to have systems of zero-emission in place, such as in industrial parks with full waste treatment, including biogas digestion and processing of manure for use as fertilizer. With current technology these systems will be costly and energy-intensive, but bio-gas, where technology is improving fast, might be an attractive option.

In parallel, there is a need to address the environmental impacts associated with production of grain, oil and protein feed. Feed is usually produced in intensive agriculture, and the principles and instruments that have been developed to control environmental issues there need to be widely applied. They include integrated pest management, and soil management and fertilization plans. In parallel, to reduce pressure on marine capture fisheries, the sector needs to develop alternatives to the use of fishmeal as feed, for example by using synthetic amino acids.

The shift to intensive production systems is accompanied by increasing size of operation, driven by economies of scale. Despite an overall growth of the sector, this is only achieved at the cost of pushing numerous small- and middle-scale producers and other agents out of business. The trend is observed in all countries following the path of intensification: in the EU and North America from as early as the 1960s, and in emerging economies since the 1980s and 1990s. This trend raises social issues of rural emigration and wealth concentration. Diversification within and outside the agricultural sector and social safety nets are some of the policies developed to address these issues.

Reorienting extensive grazing towards provision of environmental services

Grazing systems need to intensify, in those areas where the agro-ecological potential so permits, in particular for dairy production, and where nutrient balances are still negative.

In many OECD countries, excess nutrient loading is a major issue in grass-based dairy farming. Reductions in the number of livestock have been imposed, sometimes with quite positive results.

However, the vast majority of extensive grazing lands are of low productivity. Grazing occupies 26 percent of the terrestrial surface but the contribution that extensive grazing systems make to total meat production is very small with less than 9 percent of total meat supply. In areas with little potential for intensification, extensive grazing systems currently provide little in terms of productive output and have high costs in terms of environmental damage (water flows, soil losses, carbon, biodiversity).

In a world with more than 9 billion people by 2050, most of whom will be more affluent and therefore will demand environmental services, it is doubtful that these little productive extensive systems will survive, unless they include the provision of environmental services as an important, and perhaps predominant, purpose. These systems need to be re-oriented towards adding environmental service provision, rather than mere production or subsistence. This can be facilitated by payments for environmental services or other incentives to enable livestock producers to make the transition.

The central argument here is that the value of marginal land is changing and that this change will accelerate. In the past, livestock occupied vast territories because there was no possible alternative use, i.e. the land had no opportunity costs; this made marginally productive activities, such as extensive grazing, profitable.

Water-related services will likely be the first to grow significantly in importance in future, with local service provision schemes the first to be widely applied. With suitable incentives, grazers will agree to reduce and more carefully manage grazing pressure, and in certain sensitive areas to abandon grazing activities altogether.

Biodiversity-related services (e.g. species and landscape conservation) are more complex to manage, because of major methodological issues in the valuation of biodiversity, but they could find a ready uptake where they can be financed through tourism revenues. This will not be confined to rich countries. Recent examples of sharing of benefits from wildlife in Africa and elsewhere demonstrate that tourism revenues can be used to help grazers to co-habit with wildlife. Care needs to be taken that such payments for biodiversity extend beyond the "attractive" species -mammals and other species interesting to tourists - and include biodiversity at large.

Carbon sequestration services, through adjustments in grazing management or abandonment of pastures, will also be difficult, but given the potential of the world's vast grazing lands to sequester large amounts of carbon and to reduce emissions, mechanisms must be developed and deployed to use this potentially cost-effective avenue to address climate change. International agreements will require adaptation so they include carbon sequestration through Land Use, Land Use Change and Forestry (LULUCF) and the expansion of market mechanisms, which are emerging on an experimental and pilot basis.

As the scarcity of environmental resources increases, so does their value. When functioning market mechanisms can be devised, the demand for environmental services could out-compete livestock production in many diverse locations, in particular in more marginal areas where the stocking rate (and hence the gross revenues) would be only one-third of the global average. This is easier where land is under private property. It is more difficult where it under common property, particularly where large numbers of impoverished herders or smallholders depend on such land. This is not to say that responsible stewardship for natural resources does not exist in extensive grazing; rather, these systems have come under a series of endogenous (population growth) and exogenous (e.g. arable encroachment) pressures, resulting in growing environmental deterioration.

Grazing access will have to be restricted and managed, often in a way that makes livestock production a secondary output, and environmental services primary one. This is already happening in the Alps and other areas in Europe or North America, which are both environmentally vulnerable and precious in environmental terms. Payment for environmental services will have to occur at local, national and international level, depending on the nature of the service - water and soil conservation are local goods whereas biodiversity and carbon are global goods.

The large areas that have become degraded as a result of poor management and grazing pressure can be restored if countries realize the immense damage resulting from "laisser faire" and the equally important potential gains from a process guided more consciously by environmental considerations. The opportunities for this transition depend on the relative value of the productive potential of a given area, compared with its potential for environmental service provision (Lipper, Pingali and Zurek, 2006). The lower the agricultural productivity (e.g. poor soil, steep slope) and the higher the potential for environmental service provision (e.g. watershed protection), the easier the change. Degraded grazing areas fit the bill, particularly in the more humid and hilly or mountainous areas of developing countries, but making the change will still require appropriate institutional arrangement for sellers and buyers of environmental services, at the local, national and global scale. Hence, developing such schemes needs to be given priority.

Suggesting a shift from current "extractive" grazing practices to environmental service-oriented grazing raises questions of paramount importance: how to share benefits from environmental services and how to deal with the poor who currently derive their livelihoods from extensive livestock? Their numbers are considerable. Livestock provides an important source of livelihood in poor countries. In Mauritania (where it provides 15 percent of GDP), the Central African Republic (21 percent) and Mongolia (25 percent). However, this does not automatically imply that the livestock sector provides an avenue for poverty reduction.

Obviously there is no silver bullet. Alternative employment generation and out-migration and social safety nets are some of the more obvious policy needs. Arguably, the establishment of social safety nets for these populations, can be seen as an international obligation, especially in countries where the economic potential for other sectors is also limited, and where global assets such as biodiversity or climate are concerned. Such measures, combined with payments for environmental services, could facilitate the transition from mining of marginal grazing lands to a more sustainable use of these vast areas.

7.3 The challenge ahead

Livestock is a sector of striking contrasts. Though of modest economic importance, it still has overwhelming social importance in many developing countries, and still commands significant political clout in many developed countries. It causes considerable environmental damage in terms of climate change and air pollution, water supply and quality, and biodiversity. This is in stark contrast to the positive effects in waste recycling and conservation of non-renewable resources that characterized most mixed farming following the Agricultural Revolution. At the same time, livestock-dependent livelihoods of people living in, or at the margins of, poverty, are threatened.

A major outcome of this assessment is that, compared to its economic performance, the environmental impacts of the livestock sector are not being adequately addressed, despite the fact that major reductions in impact could be achieved at reasonable cost. The problem therefore lies mainly with institutional and political obstacles, and the lack of mechanisms to provide environmental feed-back, ensure that externalities are accounted for and embed the stewardship of common property resources into the sector.

Why is this so? First, civil society seems to have an inadequate understanding of the scope of the problem. Perhaps even among the majority of environmentalists and environmental policy-makers, the truly enormous impact of the livestock sector on climate, biodiversity and water is not fully appreciated. Hopefully, this assessment will help to remedy that situation.

Second, environmentally motivated action by civil society usually focuses on the functions and protection of specific ecosystems. As we have seen, the mobility of the livestock industry allows its relocation without major problems becoming apparent. However, the pressure on the environment is usually shifted elsewhere, and manifests itself in different forms. For example, intensification may reduce pressure on grazing lands but increase pressure on waterways.

Third, and related to this, is the complexity of livestock-environment interactions, and their many manifestations, make concerted actions more difficult. That is also true of many environmental issues and is a major reason why environmental policy-making lags behind other areas.

Finally, the livestock sector is driven by other policy objectives. Decision-makers find it difficult to address economic, social, health and environmental objectives simultaneously. The fact that so many people depend on livestock for their livelihoods limits the available options to policy-makers, and involves difficult and political sensitive decisions on trade-offs.

Despite these difficulties, the impact of livestock on the local and global environment is so significant that it needs to be addressed with urgency. Information, communication and education will play critical roles towards the promotion of an enhanced willingness to act.

Consumers, because of their strong and growing influence in determining the characteristics of products, will likely be the main source of commercial and political pressure to push the livestock sector into more sustainable forms. Major progress has been made in the fisheries and forestry sectors in eco-labelling of sustainably harvested fish and forest products. Eco-labels such as those of the Marine and Forest Stewardship Councils have already gained consumers' interest. This has not yet emerged in the meat and milk sector. Institutions are urgently required for the appropriate certification and labelling to guide consumers in discerning between products produced in an environment-friendly way and others. The development and application of environmental standards critically relies on functioning institutions that need to include specific environmental challenges of the livestock sector.

Many of the negative environmental impacts occur in an institutional void, without adequate institutions either to monitor the scale of the problem or to deal with it. Traditional institutions, that used to regulate access to common property resources, have become ineffective or disappeared altogether. These now need to be revived and adapted. Meanwhile, modern institutions, which would regulate the problems, are not emerging fast enough. The surge in industrial production in Asia and Latin America has not been accompanied by a concomitant upgrading of environmental regulations and related enforcement. This has led to much of the unparalleled environmental damage that is currently occurring.

Environmental damage is "traded" in the form of feed and livestock products, without the real costs appearing in the trade balance (Galloway *et al*., 2006). Appropriate institutions are required to establish more appropriate pricing mechanisms that truly signal natural resource shortages and externalities.

Policy-makers are faced with the quandary of achieving the multiple objectives of affordable supply of high value food, food safety, livelihoods and environmental soundness in a sector that, while industrializing, is still dominated by large numbers of small-scale producers in many parts of the world. In fact, concern for family-based farming is prominent in the livestock policies of many countries.

Expecting the livestock sector to deliver on all fronts is ambitious. It will require difficult choices; the policy framework for the livestock sector, as for other areas, is characterized by a large number of trade-offs. For example, a large commercial expansion of the sector, benefiting from economies of scale and with upgraded food safety standards, creates barriers to smallholder producers. Many simply will not have the financial and technical means to compete and will be forced out of business. Likewise, distortions and externalities can be corrected but the costs of higher input prices and environmental controls will have to be passed on to the consumer, in the form of higher prices for meat, milk and eggs. As we have seen, the world's rapidly growing middle class might be willing to pay the higher costs.

Current trends of structural change imply the likely and probably accelerating exit of smallholder livestock producers in developing countries as well as developed. This trend is likely to persist even where suitable institutional mechanisms, such as cooperatives and contract farming, can be used to connect smallholders to the growing and modernizing agri-business. Such mechanisms are important for buffering the social impact of structural change. However, many poor people engage in livestock activities for lack of alternative rather than out of choice, the demise of smallholders may not always be bad. This is already happening in OECD countries, it is generally not regarded as a problem, and adequate employment possibilities exist outside the sector.

However, it becomes a major social problem if such employment opportunities do not exist in other sectors and social safety nets will then be required. Policies that attempt to stem the trend of structural change, in favour of small-scale or family farming, will be costly. As demonstrated by the EU's agricultural policy, they may only prolong the process and perhaps still fail. The important issue will be to find alternative options for displaced people to gain a living outside the livestock or agricultural sector.

Given the planet's finite natural resources, and the additional demands on the environment from a growing and wealthier world population, it is imperative for the livestock sector to move rapidly towards far-reaching change. The present analysis suggests four lines of action.

First, there is a need for continued efficiency gains in resource use for livestock production, on the basis of much-required price corrections for inputs, and replacing current suboptimal production with advanced production methods - at every step from feed production, through livestock production and processing, to distribution and marketing.

Second, there is a need to accept that the intensification and perhaps industrialization of livestock production is the inevitable long-term outcome of the structural change process that is ongoing for most of the sector. The key to making this process environmentally acceptable is facilitating the right location to enable waste recycling on cropland, and applying the right technology, especially in feeding and waste management. Locating industrial livestock units in suitable rural environments and not in congested peri-urban or otherwise favoured settings allows for the recycling of nutrients.

Third, extensive land-based production will continue to exist. However, grassland-based production will need to turn to the provision of environmental services as a major purpose, and probably as the most important one in vulnerable areas. It must adjust itself to deliver landscape maintenance, biodiversity protection, clean water and eventually carbon sequestration, rather than only production of conventional livestock commodities.

Last, but certainly not least, for the suggested changes to occur, there is an urgent need to develop and implement effective policy frameworks at the local, national and international level. This will need to be established with a strong political commitment, based on a civil society that needs to be more aware of the environmental risks of continuing "business as usual."

The livestock sector is responsible for a significant share of environmental damage. With these changes, undertaken with an appropriate sense of urgency, the sector can make a very significant contribution to reducing and reversing environmental damage.

References

References

Ackerman, F., Wise T.A., Gallagher, K.P., Ney, L. & Flores, R. 2003. *Free trade, corn and the environment: Environmental impacts of US–Mexico corn trade under NAFTA.* Global Development and Environment Institute, Working Paper No. 03-06.

ADB. 2001. *Fire, smoke, and haze – the ASEAN response strategy.* Edited by S. Tahir Qadri. Asian Development Bank. Manila, Philippines. p. 246

Ahmed, M. 2000. Water pricing and markets in the Near East: Policy issues and options. *Water Policy*, 2: 229–242.

Ajayi, S.S. 1997. Pour une gestion durable de la faune sauvage: Le cas africain. In Etude FAO forêts , eds. *Ouvrages sur l'Aménagement Durable des Forêts*. Rome, FAO.

Alder, J. & Lugten, G. 2002. Frozen fish block: how committed are North Atlantic States to accountability, conservation and management of fisheries? *Marine Policy*, 26: 345–357.

Allan, J.A. 2001. Virtual Water – economically invisible and politically silent – a way to solve strategic water problems, *International Water and Irrigation*, 21(4): 39–41.

Altieri, M. & Pengue, W. 2006. GM soybean: *Latin America's new coloniser*. Article in *Grain* (Available at http://www.grain.org/front/).

Amon, B., Moitzi, G., Schimpl, M., Kryvoruchko, V. & Wagner-alt, C. 2002. *Methane, nitrous oxide and ammonia emissions from management of liquid manures, Final Report 2002.* On behalf of Federal Ministry of Agriculture, Forestry, Environmental and Water Management and the Federal Ministry of Education, Science and Culture Research Project No. 1107, BMLF GZ 24.002/24-IIA1a/98 and Extension GZ 24.002/33-IIA1a/00.

Anderson, D.M., Galloway, S.B. & Joseph, J.D. 1993. *Marine biotoxins and harmful algae: a national plan*. Technical Report, Woods Hole, Massachusetts, USA, Woods Hole Oceanographic Institution (WHOI) 93–02. 59 pp.

Anderson, K. & Martin, W. 2005. *Agricultural trade reform and the Doha Development Agenda*. World Bank Policy Research Working Paper 3607. World Bank, 21 February 2005.

Anderson, M. & Magleby, R. 1997. *Agricultural resources and environmental indicators, 1996–97.* Agricultural Handbook No. 712. July 1997, 356 pp.

Andreae, M.O. & Crutzen, P.J. 1997. Atmospheric aerosols: biogeochemical sources and roles in atmospheric chemistry. *Science*, 276: 1052–1057.

Andreoni, J & D. Capman. 2001. The simple analysis of the environmental Kuznets curve. *Journal of Public Economics* 80(2): 269–277.

Animal Info. 2005. *Information on endangered mammals*, (Available at http://www.animalinfo.org/index. Accessed July, 2005).

Archer, S., Schimel, D.S. & Holland, E. A. 1995. Mechanisms of shrubland expansion: land use, climate or CO2? *Climatic Change*, 29: 91–99.

ARKive. 2005. *Globally endangered chapter*, (Available http://www.arkive.org/species/GES/. Accessed July, 2005).

Artaxo, P., Martins, J.V., Yamasoe, M.A., Procópio, A.S., Pauliquevis, T.M., Andreae, M.O., Guyon, P., Gatti, L.V. & Leal, A.M.C. 2002. Physical and chemical properties of aerosols in the wet and dry seasons in Rondonia, Amazonia. *Journal of Geophysical Research*, 107 (D20): 8081–8095.

Arthur, J.A. & Albers, G.A.A. 2003. Industrial perspective on problems and issues associated with poultry breeding. In W.M. Muir, *Poultry genetics, breeding and biotechnology*.

Asad, M., Azevedo, L.G., Kemper, K.E. & Simpson, L.D. 1999 *Management of water resources: Bulk water pricing in Brazil*. World Bank Technical Paper No. 432.

Asner, G.P., Borghi, C.E. & Ojeda, R.A. 2003. Desertification in central Argentina: Changes in ecosystem carbon and nitrogen from imaging spectroscopy. *Ecological Application*, 13(3): 629–648.

Asner, G.P., Elmore, A.J., Olander, L.P., Martin, R.E. & Harris, A.T. 2004. Grazing systems, ecosystem responses, and global change. *Annual review of environment and resources*, 29: 261–299.

Atwill, E.R. 1995. *Microbial pathogens excreted by livestock and potentially transmitted to humans through water.* Davis, USA, Veterinary Medicine Teaching and Research Center School of Veterinary Medicine, University of California.

Baillie, J.E.M., Hilton-Taylor, C. & Stuart, S.N. eds., 2004. 2004 IUCN red list of threatened species. A global species assessment. Gland, Switzerland and Cambridge, UK, IUCN.

Baker, B., Barnett, G. & Howden, M. 2000. *Carbon sequestration in Australia's rangelands. Proceedings workshop Management options for carbon sequestration in forest, agricultural and rangeland ecosystems*, CRC for Greenhouse Accounting, Canberra.

Ballan, E. 2003. De participation en conflit: la décision partagée à l'épreuve des faits dans la moyenne vallée du Zambèze. In Rodary E., Castellanet C. & Rossi G., eds. *Conservation de la nature et développement, l'intégration possible?* Paris: Karthala & GRET, 225–237.

Bari, F., Wood, M.K. & Murray, A.L. 1993. Livestock grazing impacts on infiltration rates in a temperate range of Pakistan. *Journal of Range Management*, 46: 367–372.

Barraud, V., Saleh, O.M. and Mamis, D. 2001. *L'élevage transhumant au Tchad Oriental*. Tchad: Vétérinaires Sans Frontières.

Barrios, A. 2000. *Urbanization and water quality.* CAE Working Paper Series. WP00-1. American Farmland Trust's Center for Agriculture in the Environment, DeKalb, Ill.

Barrow, C.J. 1991. *Land degradation: Development and breakdown of terrestrial environments*. UK, Cambridge University Press. 313 pp.

Barrow, N.J. & Lambourne, L.J. 1962. Partition of excreted nitrogen, sulphur, and phosphorus between the faeces and urine of sheep being fed pasture. *Australian Journal of Agricultural Research*, 13(3): 461–471.

Batjes, N.H. 2004. Estimation of soil carbon gains upon improved management within croplands and grasslands of Africa. *Environment, Development and Sustainability*, 6:133–143.

Behnke, R. 1997. Range and Livestock Management in the Etanga Development Area, Kunene Region. Progress Report for the NOLIDEP Project. Windhoek, Namibia, Ministry of Agriculture, Water and Rural Development.

Bellamy, P.H., Loveland, P.J., Bradley, R.I, Lark, R.M. & Kirk, G.J.D. 2005. Carbon losses from all soils across England and Wales 1978–2003. *Nature*, 437: 245–248.

Bellows, B. 2001. *Nutrient cycling in pastures-livestock systems guide*. Fayetteville, Arizona, USA, ATTRA – National Sustainable Agriculture Information Service.

Belsky, A.J., Matzke A. & Uselman S. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. *Journal of Soil and Water Conservation*, 54: 419–431.

Benoît, M. 1998. *Statut et usages du sol en périphérie du parc national du «W» du Niger.* Paris, Niamey, ORSTOM.

Berg, C. 2004. *World fuel ethanol analysis and outlook*, (Available at http://www.distill.com/World-Fuel-Ethanol-A&O-2004.html).

Bernstein S. 2002. Freshwater and human population: A global perspective. In Karen Krchnak, ed., *Human population and freshwater resources: US cases and international perspective*, New Haven, USA, Yale University. 177 pp.

Bernués, J.L. Riedel, M.A. Asensio, M. Blanco, A. Sanz, R.R. & Casasús I. 2005. An integrated approach to studying the role of grazing livestock systems in the conservation of rangelands in a protected natural park (Sierra de Guara, Spain). *Livestock production science*, 96(1): 75–85.

Biggs, R., Bohensky, E., Desanker, P. V., Fabricius, C., Lynam, T., Misselhorn, A., Musvoto, C., Mutale, M., Reyers, B., Scholes, R.J., Shikongo, S. & van Jaarsveld, A.S. 2004. *Nature supporting people: the Southern Africa Millenium Ecosystem Assessment*. Pretoria, Council for Scientific and Industrial Research.

Binot, A., Castel, V. & Caron, A. 2006. *The wildlife-livestock interface in sub-Saharan Africa*. Sécheresse, June 2006.

BirdLife International. 2005. *Species factsheets*. (Downloaded from http://www.birdlife.org. Accessed July 2005).

Black, R. 2006. Public says 'no' to badger cull. BBC News, (Available at http://news.bbc.co.uk/2/hi/science/nature/5172360.stm).

Bolin, B., Degens, E.T., Kempe, S. & Ketner, P. eds. 1979. SCOPE 13 *The global carbon cycle*. Scientific Committee On Problems of the Environment (SCOPE), (Available at http://www.icsu-scope.org/downloadpubs/scope13/).

Bolin, B., Crutzen, P.J., Vitousek, P.M., Woodmansee, R.G., Goldberg, E.D. & Cook, R.B. 1981. *An overview of contributions and discussions at the SCOPE workshop on the interaction of biogeochemical cycles*, Örsundsbro, Sweden, 25–30 May 1981, (Available at www.icsu-scope.org/downloadpubs/scope21/chapter01.html).

Bosworth, B., Cornish, G., Perry, C. & van Steenbergen, F. 2002. *Water charging in irrigated agriculture Lessons from the literature*.

Bouman, B.A.M., Plant, R.A.J. & Nieuwenhuyse, A. 1999. Quantifying economic and biophysical sustainability trade-offs in tropical pastures. *Ecological Modelling*, 120(1): 31–46.

Bourgeot, A. and Guillaume, H. 1986. *Introduction au nomadisme: mobilité et flexibilité?* Bulletin de liaison No. 8. ORSTOM.

Bouwman, A.F. 1995. *Compilation of a global inventory of emissions of nitrous oxide*. Ph.D. Thesis, Agricultural University, Wageningen.

Bouwman, A.F., Lee, D.S., Asman, W.A.H., Dentener, F.J., Van Der Hoek, K.W. & Olivier J.G.J. 1997. A global high-resolution emission inventory for ammonia, *Global Biogeochemical Cycles*, 11(4): 561–587.

Bouwman, A.F., & van Vuuren, D.P. 1999. *Global assessment of acidification and eutrophication of natural ecosystems*. RIVM report 402001012. Bilthoven, the Netherlands National Institute of Public Health and the Environment (RIVM). p. 51.

Bowman, R.L., Croucher, J.C., Picard, M.T., Habib, G., Basit Ali Shah, Wahidullah, S., Jabbar, G., Ghufranullah, Leng, R.A., Saadullah, M., Safley, L.M., Cassada, M.E., Woodbury, J.W. & Perdok, H.B. 2000. *Global impact domain: Methane emissions*. Working Document LEAD, Rome, (Available at www.fao.org/WAIRDOCS/LEAD/X6116E/X6116E00.HTM).

Boyd, J.W., Caballero, K. & Simpson, R.D. 1999. *The law and economics of habitat conservation: Lessons from an analysis of easement acquisitions*. Discussion Paper 99–32. Resources for the Future, Washington, DC.

Breman, H. & de Wit, C.T. 1983. Rangeland productivity and exploitation in the Sahel. *Science*, 221(4618): 1341–1347.

British Columbia Ministry of Forests. 1997. *Remedial measures primer*. Forest Practices Branch, Forest Service British Columbia, Canada, (Available at http://www.for.gov.bc.ca/hfd/pubs/Docs/Fpb/RMP-01.htm).

Bromley, D.W. 2000. Property regimes and pricing regimes in water resource management. In Ariel Dinar, ed., *The political economy of water pricing reforms*. New York, USA, Oxford University Press. pp. 37–47.

Brown, A.E., Zhang, L., McMahon, T.A., Western, A.W. & Vertessy, R.A. 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation *Journal of Hydrology*, 310(1-4): 28–61.

Brown, I.H., Londt, B.Z., Shell, W., Manvell, R.J., Banks, J., Gardner, R., Outtrim, L., Essen, S.C., Sabirovic, M., Slomka, M. & Alexande, D.J. 2006. *Incursion of H5N1 'Asian lineage virus' into Europe: source of introduction?* FAO/OIE International Scientific Conference on Avian Influenza and Wild Birds.

Brown, J.H. 1989. Patterns, modes and extents of invasions by vertebrates. In: Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Rejmànek,M., and Williamson M., eds., *Biological invasions, a global perspective*, SCOPE 37 – Scientific Committee on Problems of the Environment. Published by John Wiley & Sons Ltd. 506 pp.

Brown, L.R. 2002. *Water deficits growing in many countries water shortages may cause food shortages*. Earth Policy Institute, 6 August 2002–11, (Available at www.earth-policy.org/Updates/Update15.htm).

Bruijnzeel, L.A. 2004. Hydrological functions of tropical forests: not seeing the soil for the trees? *Agriculture, Ecosystems & Environment*, 104(1): 185–228.

Bryant, D., Burke, L., McManus, J. & Spalding, M. 1998. *Reefs at risk: A map-based indicator of potential threats to the world's coral reefs*. pp. 56.

Bull, W.B. 1997. Discontinuous ephemeral streams. *Geomorphology*, 19(3–4): 227–276.

Bureau of Land Management. 2005. Resource Management Plan. Bakersfield Field Office (Available at http://www.ca.blm.gov/bakersfield/bkformp/rmpcontents.html. Last updated:07/20/05).

Buret, A., deHollander, N., Wallis, P.M., Befus, D. & Olson, M.E. 1990. Zoonotic potential of giardiasis in domestic ruminants. *The Journal of Infectious Diseases*, 162: 231–237.

Burton, C.H. 1997. *Manure management – treatment strategies for sustainable agriculture*. Silsoe Research Institute, Wrest Park, Silsoe, Bedford. UK. p. 196.

Burton, C.H. & Turner, C. 2003. Manure management – treatment strategies for sustainable agriculture. 2nd Edition. Wrest Park, Silsoe, Bedford, UK. Silsoe Research Institute, p. 451.

Butt, T.A., McCarl, B.A., Angerer, J., Dyke, P.R. & Stuth, J.W. 2004. *Food security implication of climate change in developing countries: findings from a case study in Mali*. USA, Texas A&M University.

Byerlee, D., Alex, G. & Echeverría, R.G. 2002. The evolution of public research systems in developing countries: Facing new challenges. In: Byerlee, D. and Echeverria, R.G., eds. *Agricultural research policy in an era of privatization*. CAB International 2002.

Byers, B.A. 1997. *Environmental threats and opportunities in Namibia: A comprehensive assessment*, Directorate of Environmental Affairs-Ministry of Environment and Tourism.

California Trout. 2004. *Grazing reform overview*, (Available at http://www.caltrout.org/index.html).

Canadian Animal Health Institute. 2004. *Hormones: A safe, effective production tool for the Canadian beef industry*. CAHI factsheet, (Available at http://www.cahi-icsa.ca/pdf/Beef-Hormones-Factsheet.pdf).

Cantagallo, J.E., Chimenti, C.A. & Hall, A.J. 1997. Number of seeds per unit area in sunflower correlates well with a photothermal quotient. *Crop Science*, 37: 1780–1786.

Carlyle, G.C. & Hill, A.R. 2001. Groundwater phosphate dynamics in a river riparian zone: effects of hydrologic flowpaths, lithology and redox chemistry. *Journal of Hydrology*, 247 (3–4): 151–168.

Carney, J.F., Carty C.E. & Colwell R.R. 1975. Seasonal occurrence and distribution of microbial indicators and pathogens in the Rhode river of Chesapeake Bay. *Applied and Environmental Microbiology*, 30(5): 771–780.

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N. & Smith, V.H. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications*, 8(3): 559–568.

Carvalho, G., Moutinho, P., Nepstad, D., Mattos, L. & Santilli, M. 2004. An Amazon perspective on the forest–climate connection: Opportunity for climate mitigation, conservation and development? *Environment, Development and Sustainability*, 6(1–2): 163–174.

CAST. 1999. *Animal Agriculture and Global Food Supply*. Council for Agricultural Science and Technology (CAST) ISBN 1-887383-17-4, July 1999, 92 pp.

Castel V. 2004. *Valeurs et Valorisation des ressources de la biodiversité: Quel Bilan? Quelles perspectives pour les éleveurs?* Introductive document of theme 1 and 2. Electronic conference of the LEAD/FAO francophone platforme: Cohabitation ou compétition entre la faune sauvage et les éleveurs... Où en est-on aujourd'hui? Faut-il changer d'approche? Organized by LEAD and CIRAD.

Castel, V. 2005. Synthèse des débats du Thème No. 1 «Valeurs et Valorisation des ressources de la biodiversité: Quel Bilan? Quelles perspectives pour les éleveurs?» de la 2ème Conférence électronique de la Plateforme francophone LEAD (FAO): *Cohabitation ou compétition entre la faune sauvage et les éleveurs... Où en est-on aujourd'hui? Faut-il changer d'approche?* Organized by LEAD and CIRAD, 2005.

Cattoli, G. and Capua, I. 2006. *A diagnostic approach to wild bird surveillance and environmental sampling*. FAO/OIE International Scientific Conference on Avian Influenza and Wild Birds.

Cederberg, C. & Flysjö, A. 2004. *Life cycle inventory of 23 dairy farms in south-western Sweden*. SIK report No. 728. p. 59.

Cerejeira, M.J., Viana, P., Batista, S., Pereira, T., Silva, E., Valerio, M.J., Silva, A., Ferreira M. & Silva-Fernandes A.M. 2003. Pesticides in Portuguese surface and groundwaters. *Water Research*, 37(5):1055–1063.

Chamberlain, D.J. & Doverspike, M.S. 2001. Water tanks protect streambanks. *Rangelands*, 23(2): 3–5.

Chameides, W.L. & Perdue, E.M. 1997. *Biogeochemical cycles: a computer-interactive study of earth system science and global change*. New York, USA, Oxford University Press.

Chapagain, A.K. & Hoekstra, A.Y. 2003. *Virtual water flows between nations in relation to trade in livestock and livestock products*. Value of Water Research Report Series No. 13. UNESCO-IHE.

Chapagain, A.K. & Hoekstra, A.Y. 2004. *Water footprints of nations*. Volume 1: Main Report. Value of Water Research Report Series No. 16. UNESCO-IHE. p. 76. (Available at http://www.waterfootprint.org).

Chapman, E.W. & Ribic, C.A. 2002. The impact of buffer strips and stream-side grazing on small mammals in southwestern Wisconsin. *Agriculture, Ecosystems and Environment*, 88: 49–59.

Chauveau, J.P. 2000. Question foncière et construction nationale en Côte d'Ivoire. *Politique Africaine*, 78: 94–125.

Child, B. 1988. The economic potential and utilization of wildlife in Zimbabwe. *Rev. Sci. tech.*, 1988.

Chohin-Kuper, A., Rieu, T. & Montginoul, M. 2003. *Water policy reform: Pricing water, cost recovery, water demand and impact on agriculture. Lessons from the Mediterranean experience*.

Christensen, V., Guenette, S., Heymans, J.J., Walters, C.J., Watson, R., Zeller, D. & Pauly, D. 2003. Hundred-year decline of North Atlantic predatory fishes. *Fish and Fisheries*, 4(1): 1–24.

Clark Conservation District. 2004. *Healthy riparian areas*, (Available at http://clark.scc.wa.gov/Page7.htm, accessed in 2004).

Cochrane, M.A. & Laurance, W.F. 2002. Fire as a large-scale edge effect in Amazonian forests. *Journal of Tropical Ecology*, 18: 311–325.

Collins R. & Rutherford K. 2004. Modelling bacterial water quality in streams draining pastoral land. *Water Research*, 38(3): 700–712.

Conceição, M.A.P., Durão, R.M.B., Costa, I.M.H., Castro, A., Louzã, A. C. & Costa, J.C. 2004. Herd-level seroprevalence of fasciolosis in cattle in north central Portugal. *Veterinary Parasitology*, 123 1–2: 93–103.

Convers, A. 2002. *Etat des lieux spatialisé et quantitative de la transhumance dans la zone périphérique d'influence du parc national du W (Niger)*. Rapport, CIRAD EMVT, 2002.

Correll, D.L. 1999. Phosphorus: a rate limiting nutrient in surface waters. *Poultry Science*, 78(5): 675–682.

Costa, J.L., Massone, H., Martnez, D., Suero, E.E., Vidal, C.M. & Bedmar F. 2002. Nitrate contamination of a rural aquifer and accumulation in the unsaturated zone. *Agricultural Water Management*, 57(1): 33–47.

Costales, A., Gerber, P. & Steinfeld, H. 2006. Underneath the livestock revolution. *Livestock Report*, 2006. Rome, FAO.

Crutzen, P.J. & Andreae, M.O. 1990. Biomass burning in the tropics: impact on atmospheric chemistry and biogeochemical cycles. *Science*, 250 (4988): 1669–1678.

Crutzen, P.J. & Goldammer, J.G. 1993. *Fire in the environment: The ecological, atmospheric, and climatic importance of vegetation fires*. Dahlem Konferenz 15–20 March 1992, Berlin), ES13, Chichester, UK, Wiley. 400 pp.

Cumming, D. H. M. 2005. Wildlife, livestock and food security in the South-East Lowveld of Zimbabwe. Pages 4146 in Proceedings of the Southern and East African Experts Panel on Designing Successful Conservation and Development Interventions at the Wildlife/Livestock Interface: Implications for Wildlife, Livestock and Human Health. Gland, Switzerland, IUCN Occasional Paper. International Union for the Conservation of Nature and Natural Resources.

d'Antonio, C.M. 2000. Fire, plant invasions and global changes, In: *Invasive Species in a Changing World*, edited by H.A. Mooney and R.J. Hobbs. Washington, DC, Island Press. pp 65–94

Dalla Villa, R., de Carvalho Dores, E.F., Carbo. L. & Cunha, M.L. 2006. Dissipation of DDT in a heavily contaminated soil in Mato Grosso, Brazil. *Chemosphere*, 64(4): 549–54.

Daniel, T.C., Sharpley, A.N., Edwards, D.R., Wedepohl, R. & Lemunyon, J.L. 1994. Minimizing surface water eutrophication from agriculture by phosphorous management. *Journal of Soil and Water Conservation*, 49(2): 30.

Darlington, P. J., Jr. 1943. Carabidae of mountains and islands: data on the evolution of isolated faunas, and on atrophy of wings. *Ecological Monographs* 13, 37–61.

David, H.M. 2005. Wildlife, livestock and food security in the South East of Zimbabwe. In *conservation and development interventions at the wildlife/livestock interface implications for wildlife, livestock and human health*. Edited and compiled by Steven A. Osofsky. Proceedings of the Southern and East African Experts Panel on Designing Successful Conservation and Development Interventions at the Wildlife/Livestock Interface: Implications for Wildlife, Livestock and Human Health, AHEAD (Animal Health for the Environment And Development), 14th and 15th September, 2003. Occasional Paper of the IUCN Species Survival Commission No. 30. pp. 41–46.

de Haan, C.H., Steinfeld, H. & Blackburn, H. 1997. *Livestock and the environment: Finding a balance*. Suffolk, UK, WRENmedia.

de Haan, C.H., Schillhorn van Veen, T.W., Brandenburg, B., Gauthier, J., Le Gall, F., Mearns, R. & Siméon, M. 2001. *Livestock development, implications for rural poverty, the environment, and global food security*. Washington, DC, World Bank.

De la Rosa, D., Moreno, J. A., Mayol, F. & Bonson, T. 2000. Assessment of soil erosion vulnerability in western Europe and potential impact on crop productivity due to loss of soil depth using the ImpelERO model. *Agriculture, Ecosystems and Environment*, 1590: 1–12.

de Wit, J., van Keulen, H., van der Meer, H.G. & Nell, A.J. 1997. Animal manure: asset or liability? *World Animal Review* 88-1997/1, (Available at www.fao.org/docrep/w5256t/W5256t05.htm).

Delgado, C. and Narrod, C. 2002. *Impact of changing market forces and policies on structural change in the livestock industries of selected fast-growing developing countries*. Final research report of phase I – project on livestock industrialization, trade and social-health-environment impacts in developing countries.

Delgado, C., Narrod, C.A. and Tiongco, M.M. 2003. *Policy, technical, and environmental determinants and implications of the growing scale of livestock farms in four fast–growing developing countries*. International Food Policy Research Institute Washington, DC.

Delgado, C., Rosegrant, M., Steinfeld, H., Ehui, S. & Courbois, C. 1999. Livestock to 2020: *The next food revolution*. Food, Agriculture, and the Environment Discussion Paper 28. Washington, DC, IFPRI/FAO/ILRI (International Food Policy Research Institute/FAO/International Livestock Research Institute).

Delgado, C., Wada, N., Rosegrant, M.W., Meijer, S. & Mahfuzuddin, A. 2003. *Fish to 2020: supply and demand in changing global markets*. Washington, DC, the International Food Policy Research Institute and the WorldFish Center.

Delgado, C., Narrod, C.A. & Tiongco, Marites M. 2006. *Determinants and implications of the growing scale of livestock farms in four fast–growing developing countries*. Washington, DC, International Food Policy Research Institute.

Department of the Environment, Sport & Territories. 1993. *Biodiversity and its value.* Biodiversity Series, Paper No. 1. Biodiversity Unit, The Department of the Environment, Sport and Territories of the Commonwealth of Australia.

Devendra, C. & Sevilla, C.C. 2002. Availability and use of feed resources in crop–animal systems in Asia, *Agricultural Systems*, 71(1): 59–73.

Devine. R. 2003. La consommation des produits carnés. *INRA Prod. Anim.*, 16(5): 325–327

Di Castri, F. 1989. History of biological invasions with special emphasis on the Old World. In: Drake, J.A., H.A. Mooney, F. di Castri, R.H. Groves, F.J. Kruger, M. Rejmànek, and M. Williamson, eds., *Biological invasions, a global perspective*, SCOPE 37 – Scientific Committee on Problems of the Environment. John Wiley & Sons Ltd. p. 506.

Di Tomaso, J.M. 2000. Invasive weeds in rangelands: Species, impacts, and management. *Weed Science*, 48(2): 255–265.

Di, H.J. & Cameron, K.C. 2003. Mitigation of nitrous oxide emissions in spray-irrigated grazed grassland by treating the soil with dicyandiamide, a nitrification inhibitor. *Soil use and management*, 19(4), 284–290.

Diamond, J. and Shanley, T. 1998. *Infiltration rate assessment of some major soils*. Wexford, UK, Johnstown Castle Research Centre.

Dinda, S. 2005. A theoretical basis for the environmental Kuznets curve. *Ecological Economics* 53 (2005) 403–413.

Dompka, M.V., Krchnak, K.M. & Thorne, N. 2002. Summary of experts' meeting on human population and freshwater resources. In Karen Krchnak, ed., *Human Population and Freshwater Resources: U.S. Cases and International Perspective*, Yale University, New Haven, USA. 177 pp.

Donner, S.D. 2006. Surf or turf: A shift from feed to food cultivation could reduce nutrient flux to the Gulf of Mexico. *Global Environmental Change*, in press.

Douglas, J.T. & Crawford, C.E. 1998. Soil compaction effects on utilization of nitrogen from livestock slurry applied to grassland Source. *Grass and Forage Science*, 53(1): 31–34.

Dregne, H., Kassa, M. & Rzanov, B. 1991. A new assessment of the world status of desertification. *Desertification Control Bulletin*, 20, 6–18.

Dregne, H.E. 2002. Land degradation in dry lands. *Arid land research and management*, 16: 99–132.

Dregne, H.E. & Chou, N.T. 1994. Global desertification dimensions and costs. In H.E. Dregne, ed. *Degradation and restoration of arid lands*. Lubbock; USA; Texas Technical University.

East Bay Municipal Utility District. 2001. *East Bay watershed range resource and management plan* (RRMP). East Bay Municipal Utility District, Watershed and Recreation Division, (Available at http://www.ebmud.com/water_&_environment/environmental_protection/east_bay/range_resource_management_plan/).

Eckard, R., Dalley, D. & Crawford, M. 2000. *Impacts of potential management changes on greenhouse gas emissions and sequestration from dairy production systems in Australia*. Proceedings workshop "Management Options for Carbon Sequestration in Forest, Agricultural and Rangeland Ecosystems", CRC for Greenhouse Accounting, Canberra.

Els, A.J.E. & Rowntree, K.M. 2003. *Water resources in the savannah regions of Botswana*. EU INCO/UNEP/SCOPE Southern African Savannas Project.

Engels, C.L. 2001. *The effect of grazing intensity on rangeland hydrology*. NDSU Central Grasslands Research Extension Center. Available at www.ag.ndsu.nodak.edu/streeter/2001report/Chad_engels.htm.

English, W.R., Wilson, T. & Pinkerton, B. (No date). *Riparian management handbook for agricultural and forestry lands*. College of Agriculture, Forestry and Life Sciences, Clemson University, Clemson.

EPICA community members. 2004. Eight glacial cycles from an Antarctic ice core. *Nature*, 429. 10 June. pp. 623–628.

Estergreen, V.L., Lin, M.T., Martin, E.L., Moss, G.E., Branen, A.L., Luedecke, L.O. & Shimoda, W. 1977. Distribution of progesterone and its metabolites in cattle tissues following administration of progesterone-4–14C. *Journal of Animal Science*, 45(3): 642–651.

Eswaran, H., Lal, R. & Reich, P.F. 2001. Land degradation: an overview. In E.M. Bridges, I.D. Hannam, L.R. Oldeman, F.W.T. Pening de Vries, S.J. Scherr & S. Sompatpanit, eds., *Responses to land degradation*. Proceedings of the second International Conference on Land Degradation and Desertification, Khon Kaen, Thailand. New Delhi, Oxford Press.

European Commission. 2004. *European pasture monography and pasture knowledge base* PASK study, (Available at http://agrifish.jrc.it/marsstat/Pasture_monitoring/PASK/).

European Union.1992. Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. *Annex I: Natural habitat types of Community interest whose conservation requires the designation of special areas of conservation*. (Available at http://web.uct.ac.za/depts/pbl/jgibson/iczm/legis/ec/hab-an1.htm).

EU. 2006. *European Union web site* (Available at http://europa.eu/index_en.htm, accessed April 2006).

Falvey, L. & Chantalakhana, C. eds., 1999. *Smallholder dairying in the tropics*. International Livestock Research Institute (ILRI), Nairobi, Kenya. 462 pp.

FAO-AQUASTAT. 2004. AQUASTAT databases. FAO.

FAO. 1996. *World livestock production systems: Current status, issues and trends*, by C. Seré & S. Steinfeld. FAO Animal Production and Health Paper 127, Rome.

FAO. 1997. *Review of the state of world aquaculture*. FAO Fisheries Circular, No. 886, Rev.1., Rome.

FAO. 1999a. *The state of world fisheries and aquaculture 1998*. Rome.

FAO. 1999b. *Trade, environment and sustainable development*. Third WTO ministerial conference, Seattle, 28 November- 3 December 1999, (Available at http://www.fao.org/documents/show_cdr.asp?url_file=/DOCREP/003/X6730E/X6730E01.HTM).

FAO. 1999c. *Livestock and environment toolbox*, Livestock, Environment and Development (LEAD) Initiative. Rome. (Available at www.virtualcentre.org/en/dec/toolbox/homepage.htm).

FAO. 2000a. *Two essays on climate change and agriculture*. FAO Economic and Social Development Paper 145, (Available at http://www.fao.org/docrep/003/x8044e/x8044e00.HTM).

FAO. 2000b. Agro-ecological Zoning System, (Available at www.fao.org/ag/agl/agll/prtaez.stm).

FAO. 2002. *Fertilizer use by crop*. Joint report FAO, IFA, IFDC, IPI, PPI. 5th edition. p. 45, (Available at www.fertilizer.org/ifa/statistics/crops/fubc5ed.pdf).

FAO. 2003a. *World agriculture: Towards 2015/30*, An FAO perspective, edited by J. Bruisnsma, FAO. Rome and London, Earthscan.

FAO. 2003b. *Presentation from the Area Wide Integration Project* (LEAD), Bangkok Workshop.

FAO. 2003c. *Novel forms of livestock and wildlife integration adjacent to protected areas in Africa* – (LEAD) project document.

FAO. 2004a. *The role of soybean in fighting world hunger*. Rome, (Available at http://www.fao.org/es/esc/common/ecg/41167_en_The_role_of_soybeans.pdf).

FAO. 2004b. Carbon sequestration in dryland soils. World Soils Resources Reports 102, (Available at www.fao.org/docrep/007/y5738e/y5738e00.htm).

FAO. 2004c. Biodiversity for Food Security. World Food Day, October 2004. FAO, (Available at http://www.fao.org/wfd/2004/index_en.asp).

FAO. 2004d. *Payment schemes for environmental services in watersheds*. Regional forum, 9–12 June 2003, Arequipa, Peru. Organized by the FAO Regional Office for Latin America and the Caribbean; Santiago, Chile. FAO, Rome 2004.

FAO. 2004e. *Livestock waste management in East Asia*. Project Preparation Report, FAO, Rome.

FAO. 2005a. Special Event on Impact of Climate Change, Pests and Diseases on Food Security and Poverty Reduction. Background Document, 31st Session of the Committee on World Food Security, 23–26 May 2005. Rome.

FAO. 2005b. *State of the world's forests*. FAO, Rome. 153 pp.

FAO. 2005c. Review of the state of world marine fishery resources: Global overiew – Global production and state of the marine fishery resources. FAO Fisheries Technical Paper 457, (Available at http://www.fao.org/docrep/009/y5852e/Y5852E02.htm).

FAO. 2005d. *Livestock Sector Brief: China*. Livestock Information, Sector Analysis and Policy Branch. Animal Production and Health Division, FAO. (Available at www.fao.org/ag/againfo/resources/en/publications/sector_briefs/lsb_CHN.pdf).

FAO. 2005e. *Pollution from industrialized livestock production*. Livestock Policy Brief No.2 FAO. 2005f. Global forest resources assessment. FAO Forestry Paper No. 147. Rome, (Available at www.fao.org/forestry/site/fra/en).

FAO. 2006a. *World agriculture: towards 2030/2050, Interim Report*. Rome.

FAO. 2006b. FAO statistical databases. Rome, (Available at http://faostat.fao.org/default.aspx).

FAO. 2006c. Second report on the State of the World's Animal Genetic Resources. FAO, Rome, in press.

FAO. 2006d. *Agro-ecological zones information portal*. (Available at http://www.fao.org/AG/agl/agll/prtaez.stm), Rome.

FAO. 2006e. *Cattle ranching and deforestation*. Livestock Policy Brief No.3 Animal Production and Health Division. Rome.

FAO. 2006f. Gridded livestock of the world, (Available at http://www.fao.org/ag/AGAinfo/resources/en/glw/default.html). Rome, in press.

FAO. 2006g. Food Insecurity, Poverty and Environment Global GIS Database (FGGD) and Digital Atlas for the Year 2000, Environmental and Natural Resources Working Paper 26. Rome. In press.

FAO/IFA. 2001. *Global estimates of gaseous emissions of NH_3, NO and N_2O from agricultural land*. Rome. 106 pp.

Fargeot, C. 2004. La chasse commerciale et le négoce de la venaison En Afrique centrale forestière. Proceedings de: *La Faune Sauvage: Une Ressource Naturelle*, 6ème symposium international sur l'utilisation de la faune sauvage. Paris, France 6–9 July 2004.

Fayer, R., Santin, M., Sulaiman, I.M., Trout, J., Xunde, L., Schaefer, F.W., Xiao, L. & Lal, A.A. 2002. *Animal reservoirs, vectors, and transmission of microsporidia*. Presented at American Society of Tropical Medicine and Hygiene 51st Annual Meeting, Denver, Colorado, USA, 10–14 November 2002.

Fearnside, P.M. 2001. Soybean cultivation as a threat to the environment in Brazil. *Environmental Conservation*, 28: 23–38.

Field, L.Y., Embleton, K.M., Krause, A., Jones, D. & Childress, D. 2001. *Livestock manure handling on the farm*. University of Wisconsin-Extension, Minnesota Extension Service, and the United States Environmental Protection Agency Region 5. (Available at http://danpatch.ecn.purdue.edu/~epados/farmstead/yards/src/title.htm).

Filson, G.C. 2001. *Agroforestry extension and the Western China development strategy*. Canadian Society of Extension.

Fishmeal Information Network. 2004. *Fish meal from sustainable stocks*. (Accessed: 14 October 2005).

Flanigan, V., Shi, H., Nateri, N., Nam, P., Kittiratanapiboon, K., Lee, K. & Kapila, S. 2002. *A fluidized-bed combustor for treatment of waste from livestock operations*. Conference on the Application of Waste Remediation Technologies to Agricultural Contamination of Water Resources, Great Plains/Rocky Mountain Hazardous Substance Research Center (HSRC), Kansas State University, USA, 30 July–1 August 2002.

Florinsky, I.V., McMahon, S. & Burton, D.L. 2004. Topographic control of soil microbial activity: a case study of denitrifiers. *Geoderma*, 119(1-2): 33–53.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N. & Snyder, P.K. 2005. Global consequences of land use. *Science*, 309(5734): 570–574.

Folliott, P. 2001. *Managing arid and semi-arid watersheds: Training course in watershed management*. USA, University of Arizona.

Fouchier, R.A.M., Munster, V.J., Keawcharoen, J., Osterhaus, A.D.M.E. & Kuiken Thijs. 2006. *Virology of avian influenza in relation to wild birds*. FAO/OIE International Scientific Conference on Avian Influenza and Wild Birds.

Frank, L.G., Woodroffe, R. & Ogada, M.O. (in press). People and predators in Laikipia District, Kenya. In: Woodroffe R., Thirgood S., Rabinowitz A.R., Eds. *People and wildlife – conflict or coexistence?*

Frolking, S.E., Mosier, A.R., Ojima, D.S., Li, C., Parton, W.J., Potter, C.S., Priesack, E., Stenger, R., Haberbosch, C., Dorsch, P., Flessa, H. & Smith, K.A. 1998. Comparison of N_2O emissions from soils at three temperate agricultural sites: Simulations of year-round measurements by four models. *Nutrient Cycling in Agroecosystems*, 52(2–3): 77–105.

Galloway, J.N., Schlesinger, W.H., Levy, H., II, Michaels, A. & Schnoor, J.L. 1995. Nitrogen fixation: Anthropogenic enhancement-environmental response. *Global Biogeochemical Cycles*, 9(2): 235–252.

Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S.P., Howarth, R.W., Cowling, E.B. & Cosby, B.J. 2003. The nitrogen cascade. *Bioscience*, 53(4): 341–356.

Galloway, J.N., Dentener, F.J., Capone, D.G., Boyer, E.W., Howarth, R.W., Seitzinger, S.P., Asner, G.P., Cleveland, C.C., Green, P.A., Holland, E.A., Karl, D.M., Michaels, A.F., Porter, J.H., Townsend, A.R. & Vörösmarty, C.J. 2004. Nitrogen cycles: past, present, and future. *Biogeochemistry*, 70: 153–226.

Galloway, J., Burke, M., Bradford, E., Falcon, W., Gaskell, J., McCullough, E., Mooney, H., Naylor, R., Oleson, K., Smil, V., Steinfeld, H. & Wassenaar, T. 2006. *International trade in Meat: The tip of the pork chop*. In press.

Gate Information Services – GTZ. 2002. *Treatment of tannery waste water*, factsheet.

GDRS. 2000. *Irrigation in the basin context: The Gediz River basin study, Turkey*. IWMI and GDRS (IWMI and General Directorate of Rural Services). International Water Management Institute, Colombo, Sri Lanka (2000).

Gerber, P. & Menzi, H. 2005. *Nitrogen losses from intensive livestock farming systems in Southeast Asia: a review of current trends and mitigation options*.

Gerber, P., Chilonda, P., Franceschini, G. & Menzi, H. 2005. Geographical determinants and environmental implications of livestock production intensification in Asia. *Bioresource Technology*, 96: 263–276.

Gerber, P. 2006. Putting pigs in their place, environmental policies for intensive livestock production in rapidly growing economies, with reference to pig farming in Central Thailand. Doctoral Thesis in *Agricultural Economics*, Swiss Federal Institute of Technology, Zurich, 130 pp.

Gerber, P. & Steinfeld, H. 2006. *Regional planning or pollution control: policy options addressing livestock waste, with reference to industrial pig production in Thailand*. Submitted.

Gerber, P., Carsjens, G.J., Pak-Uthai, T. & Robinson, T. 2006. Spatial decision support for livestock policies: addressing the geographical variability of livestock production systems. *Agricultural Systems*. Submitted.

Gerlach Jr., J.D. 2004. The impacts of serial land-use changes and biological invasions on soil water resources in California, USA. Journal of Arid Environments, 57: 365–379.

GESAMP (IMO/FAO/UNESCO-IOC/WMO/WHO/IAEA/UN/UNEP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection) and Advisory Committee on Protection of the Sea. 2001. *Protecting the oceans from land-based activities – Land-based sources and activities affecting the quality and uses of the marine, coastal and associated freshwater environment*. Rep. Stud. GESAMP No. 71, 162 pp.

Gibon, A. 2005. Managing grassland for production, the environment and the landscape. Challenges at the farm and the landscape level. *Livestock Production Science*, 96(1): 11–31.

Gilbert, M., Wint, W., Slingenbergh, J. 2004. Ecological factors in disease emergence from animal reservoir. FAO AGAH, unpublished report. 39 pp.

GISD. 2006. *Global Invasive Species Database*. Accessed on line, June 2006, (Available at http://www.issg.org/database/welcome/).

Gleick, P.H. 2000. Water futures: A review of global water resources projections. In: Rijsberman, F.R. ed. 2000. *World water scenarios: Analyses*. Earthscan Publications, London, pp. 27–45.

Global Footprint Network. *The ecological footprint*, (Available at http://www.footprintnetwork.org/gfn_sub.php?content=footprint_overview).

Global Land Cover. Global Land cover 2000, (Available at http://www-gvm.jrc.it/glc2000/).

Godwin, D.C. & Miner, J.R. 1996. The potential of off-stream livestock watering to reduce water quality impacts. *Bioresource Technology*, 58(3): 285–290.

Goldewijk, K. & Battjes, J.J. 1997. *A hundred year database for integrated environmental assessments*. Bilthoven, the Netherlands, National Institute of Public Health and the Environment.

Golfinopoulos, S.K, Nikolaou, A.D., Kostopoulou, M.N., Xilourgidis, N.K., Vagi, M.C. & Lekkas, D.T. 2003. Organochlorine pesticides in the surface waters of Northern Greece. *Chemosphere*, 50(4): 507–516.

Grazing & Pasture Technology Program. 1997. *Grazing Management of Rangeland: A Watershed Perspective. The Grazing Gazette*; 8(3). Grazing and Pasture Technology Program, Regina Saskatchewan, Canada.

Gretton, P. & Salma, U. 1996. *Land degradation and the Australian agricultural industry*. Industry Commission, Australian Government Publishing Service.

Groenewold, J. 2005. *Classification and characterization of world livestock production systems* – update of the 1994 livestock production systems datasets with recent data. Unpublished report.

Hadjigeorgiou, I., Osoro, K., Fragoso de Almeida, J.P. & Molle, G. 2005. Southern European grazing lands: Production, environmental and landscape management aspects. *Livestock Production Science*, 96(1): 51–59.

Hagemeijer, W. & Mundkur, T. 2006. *Migratory flyways in Asia, Eurasia and Africa and the spread of HP H5N1*. FAO/OIE International Scientific Conference on Avian Influenza and Wild Birds.

Hahn, R.W., Olmstead, S.M. & Stavins, R.N. 2003. *Environmental regulation during the 1990s: A retrospective analysis.*

Hall, S.J. & Matson, P.A. 1999. Nitrogen oxide emissions after nitrogen additions in tropical forests. *Nature*, 400 (6740): 152–155.

Hamilton, D.W., Fathepure B., Fulhage, C.D., Clarkson, W. & Lalman, J. 2001. Treatment lagoons for animal agriculture. pp. 547–574. In *animal agriculture and the environment: national center for manure and animal waste management White Papers*. J.M. Rice, D.F. Caldwell, F.J. Humenik, eds., St. Joseph, Michigan, USA, the American Society of Agricultural and Biological Engineers.

Hanley, N., Shogren, J. & White, B. 2001. *Introduction to environmental economics*, Oxford University Press.

Harper, J., George, M. & Tate, K. 1996. *What is a watershed?* Fact Sheet No. 4: Rangeland Watershed Program; U.C. Cooperative Extension and USDA Natural Resources Conservation Service; California Rangelands Research and Information Center, Agronomy and Range Science, Davis, USA, University of California.

Harrington, G. 1994. Consumer demands: major problems facing industry in a consumer-driven society. *Meat Science*, 36: 5–18.

Harris, B.L., Hoffman, D.W. & Mazac, F.J., Jr. 2005. *TEX*A*Syst*. Water Sciences Laboratory, Blackland Research Center, Temple, Texas, (Available at http://waterhome.brc.tamus.edu/index.html).

Harris, D. & Rae, A. 2006 Agricultural policy reform and adjustment in Australia and New Zealand. In D. Blandford & B. Hill, eds. *Policy reform and adjustment in the agricultural sectors of developed countries*, Oxford, UK, CABI.

Harrison, P.F. & Lederberg, J., eds., 1998. *Antimicrobial resistance: Issues and options* 1998. Forum on Emerging Infections, Institute of Medicine. Washington, DC, National Academy Press.

Harvey, B. 2001. *Biodiversity and fisheries. Chapter 1: Synthesis report, A primer for planners.* Proceedings of the international workshop funded by UNEP and IDRC "Blue Millennium: Managing Global Fisheries for Biodiversity" Victoria, BC, 25–27 June 2001.

Harvey, J.W., Conklin, M.H. & Koelsch, R.S. 2003. Predicting changes in hydrologic retention in an evolving semi-arid alluvial stream. *Advances in Water Resources*, 26(9): 939–950.

Haynes, R.J. & Williams, P.H. 1993. "Nutrient cycling and soil fertility in the grazed pasture ecosystem." *Advances in Agronomy*, 49:119–199.

Hegarty, R.S. 1998. Reducing methane emissions through elimination of rumen protozoa. *Meeting the Kyoto Target. Implications for the Australian Livestock Industries*. P.J Reyenga. and S.M. Howden, eds. Bureau of Rural Sciences, 55–61.

Helsel, Z.R. 1992. Energy and alternatives for fertilizer and pesticide use. In R.C. Fluck, ed. *Energy in farm production*. Vol.6 in Energy in world agriculture. Elsevier, New York. pp.177–201, (Available at www.sarep.ucdavis.edu/NEWSLTR/v5n5/sa-12.htm).

Herrmann S.M., Anyamba, A. & Tucker, C.J. 2005. Recent trends in vegetation dynamics in the African Sahel and their relationship to climate. *Global Environmental Change*, 15: 394–404.

Heywood, V. 1989. Patterns, extents and modes of invasions by terrestrial plants. In: Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Rejmánek, M. and Williamson, M., 1989. *Biological invasions: A global perspective*, SCOPE 37, J. Wiley & Sons. pp 31–60

Hobbs, P.T., Reid, J.S., Kotchenruther, R.A., Ferek, R.J. & Weiss, R. 1997. Direct radiative forcing by smoke from biomass burning, *Science*, 275: 1776–1778.

Hodgson, S. 2004. Land and water – the rights interface, Livelihoods Support Programme (LSP), FAO, (Available at www.fao.org/docrep/007/j2601e/j2601e00.htm).

Hoffmann, I. & Scherf, B. 2006. Animal genetic resources – time to worry? In *Livestock report 2006*. A. McLeod ed., FAO, Rome. pp 57–74.

Hooda, P.S., Edwards, A.C., Anderson, H.A. & Miller, A. 2000. A review of water quality concerns in livestock farming areas. *The Science of the Total Environment*, 250(1–3):143–167.

Houben, P., Edderai, D. & Nzego, C. 2004 L'élevage d'aulacodes : présentation des résultats préliminaires de la vulgarisation dans trois pays d'Afrique Centrale. Proceedings de: *La Faune Sauvage : Une Ressource Naturelle*, 6ème symposium international sur l'utilisation de la faune sauvage. Paris, France, 6–9 July 2004.

Houghten, J.T., Meira Filho, L.G., Lim, B., Treanton, K., Mamaty, I., Bonduki, Y., Griggs, D.J. & Callender, B.A., eds. 1997. *Revised IPCC guidelines for national greenhouse gas inventories*. Greenhouse Gas Inventory Reference Manual, Vol. 3. UK Meteorological Office, Bracknell, UK, (Available at http://www.ipcc-nggip.iges.or.jp/public/gl/invs6c.htm).

Houghton, R. A. 1991. Tropical deforestation and atmospheric carbon dioxide. *Climatic Change*, 19(1–2): 99–118.

Hrudey. 1984. Cited by UNEP Working Group for Cleaner Production in the Food Industry, 2004. Fact Sheet 7: Food Manufacturing Series, (Available at http://www.gpa.uq.edu.au/CleanProd/Res/facts/FACT7.HTM).

Hu, Dinghuan, Reardon, T.A., Rozelle, S., Timmer, P. & Wang, H. 2004. The emergence of supermarkets with Chinese characteristics: challenges and opportunities for China's agricultural development. *Development Policy Review*, 22(5): 557–586.

Hutson, S.S., Barber, N.L., Kenny, J.F., Linsey, K.S., Lumia, D.S. & Maupin, M.A. 2004. *Estimated use of water in the United States in 2000*. US Geological Survey Circular 1268, p. 46.

Ibisch, P., Jennings, M.D. & Kreft, S. 2005. Biodiversity needs the help global change managers not museum-keepers. *Nature*, 438:156.

IFA. 2002. *Fertilizer indicators*. Second edition. International Fertilizer Industry Association, Paris. p. 20, (Available at www.fertilizer.org/ifa/statistics/indicators/ind_reserves.asp).

IFFO. 2006. International Fishmeal and Fish oil Organisation. *Industry Overview*, (Available at http://www.iffo.net/default.asp?fname=1&url=253).

Immordino, G. 2003. Looking for a guide to protect the environment: The development of the precautionary principle. *Journal of Economic Surveys*, 17(5): 629.

Institute for International Cooperation in Animal Biologics. 2004. *Cryptosporidiosis: Factsheet*. Center for Food Security and Public Health, College of Veterinary Medicine Iowa State University, (Available at http://www.cfsph.iastate.edu/Factsheets/pdfs/cryptosporidiosis.pdf).

Institute for International Cooperation in Animal Biologics. 2005. *Campylobacteriosis: Factsheet*. Center for Food Security and Public Health, College of Veterinary Medicine Iowa State University, (Available at www.cfsph.iastate.edu/Factsheets/pdfs/campylobacteriosis.pdf).

International Water Management Institute. 2000. *Projected water scarcity in 2025*, (Available at http://www.iwmi.cgiar.org/home/wsmap.htm).

IPCC. 1997. *Revised 1996 IPCC guidelines for national greenhouse gas inventories – Reference manual* (Volume 3). (Available at www.ipcc-nggip.iges.or.jp/public/gl/invs6.htm).

IPCC. 2000. *Land use, land use change and forestry*. A special report of the IPCC. Cambridge, UK, Cambridge University Press.

IPCC. 2001a. *Climate change 2001: Impacts, adaptation and vulnerability.* IPCC Third Assessment Report. UK, Cambridge University Press. 1 032 pp.

IPCC. 2001b. *Climate change 2001: The scientific basis.* Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change [J.T. Houghton, Y. Ding, D.J. Griggs, M. Noguer, P.J. van der Linden, X. Dai, K. Maskell & C.A. Johnson, eds.]. Cambridge, UK and New York, Cambridge University Press. 881 pp.

IPCC. 2002. Climate change and biodiversity. Edited by H. Gitay, A. Suárez, R. T. Watson & D. J. Dokken. IPCC Technical Paper V.

Isik, M. 2004. Environmental regulation and the spatial structure of the US dairy sector. *American Journal of Agricultural Economic*, 86(4): 949.

IUCN. 2000. *IUCN Guidelines for the prevention of biodiversity loss caused by alien invasive species*. IUCN, Gland, Switzerland.

IUCN. 2004. *The 2004 IUCN red list of threatened species*, (Available at http://www.iucn.org/themes/ssc/red_list_2004/GSAexecsumm_EN.htm. Accessed July 2005).

IUCN. 2005. Wetlands and water resources: The ongoing destruction of precious habitat. International Union for Conservation of Nature and Natural Resources, (Available at http://www.iucn.org/themes/wetlands/wetlands.html).

IUCN. 2006. *Summary statistics for globally threatened species*. IUCN-World Conservation Union, Geneva.

Jagtap, S. & Amissah-Arthur, A. 1999. Stratification and synthesis of crop-livestock production system using GIS. *GeoJournal,* 47(4): 573–582.

Jalali, M. 2005. Nitrates leaching from agricultural land in Hamadan, western Iran. *Agriculture, Ecosystems & Environment*, 110 (3–4): 210–218.

Jansen, H.G.P., Ibrahim, M.A., Nieuwenhuyse, A., 't Mannetje, L., Joenje, M. & Abarca, S. 1997. The economics of improved pasture and sylvipastoral technologies in the Atlantic Zone of Costa Rica. *Tropical Grasslands*, 31: 588–598.

Jayasuriya, R.T. 2003. Measurement of the scarcity of soil in agriculture. *Resources Policy*, 29(3–4): 119–129.

Jenkinson, D.S. 1991. The Rothamsted long-term experiments: are they still of use? *Journal of Agronomy*, 83: 2–12.

Jin, L. and Young, W. 2001. Water use in agriculture in china: importance, challenges and implications for policy. *Water Policy*, 3: 215–228.

Johansson, R.C. 2000. *Pricing irrigation water – a literature survey.* Policy Research Working Paper 2249, Washington, DC, World Bank.

Johansson, R.C., Tsur, Y., Roe, T.L., Doukkali, R. & Dinar, A. 2002. Pricing irrigation water: a review of theory and practice. *Water Policy*, 4: 173–199.

Johnson, N., Ravnborg, H.M., Westermann, O. & Probst, K. 2002. User participation in watershed management and research. *Water Policy*, 3(6): 507–520.

Kallis, G. & Butler, F. 2001. The EU water framework directive: measures and implications. *Water Policy*, 3(2): 125–142.

Kawashima, T. 2006. *Use of co-products for animal feeding in Japan*. Paper presented at a workshop "Improving total farm efficiency in swine production" held in Taiwan Province of China by the Food and Fertilizer Technology Centre and by the Taiwan Livestock Research Institute.

Ke, B. 2004 *Livestock sector in China: Implications for food security, trade and environment*. Research Center for Rural Development (RCRE).

Khaleel, R, Reddy K.R. & Overcash, M.R. 1980 Transport of potential pollutants in runoff water from land areas receiving animal wastes: a review. *Water Research*, 14(5): 421–436.

Khalil, M.A.K. & Shearer, M.J. 2005. *Decreasing emissions of methane from rice agriculture*. 2nd International Conference on Greenhouse Gases and Animal Agriculture (GGAA 2005) – Working Papers, p. 307–315.

Kijne, J.W., Barker, R. & Molden, D. 2003. *Water productivity in agriculture: Limits and opportunities for improvement*. Wallingford, UK, CABI Publishing.

King, B.S., Tietyen, J.L. & Vickner, S.S. 2000. Consumer trends and opportunities. Lexington, USA, University of Kentucky.

Kinje, J. 2001. *Water for food for sub-Saharan Africa*. A background document for the e-mail conference on "Water for Food in Sub-Saharan Africa" 15 March – 23 April 1999, Rome.

Klare, M.T. 2001. *Resource wars: the new landscape of global conflict.* New York, USA, Metropolitan Books/ Henry Holt and Company.

Klimont, Z. 2001. *Current and future emissions of ammonia in China.* Proceedings of the 10th International Emission Inventory Conference "One Atmosphere, One Inventory, Many Challenges", Denver, USA, 30 April – 3 May 2001.

Klopp, J. 2002. Can moral ethnicity trump political tribalism? The struggle for land and nation in Kenya. *African Studies* 61(2): 269–294.

Kock, R.A. 2005. What is this infamous "wildlife/ livestock disease interface?" A Review of Current Knowledge for the African Continent. In: *conservation and development interventions at the wildlife/livestock interface implications for wildlife, livestock and human health*. Edited and compiled by Steven A. Osofsky. Proceedings of the Southern and East African Experts Panel on Designing Successful Conservation and Development Interventions at the Wildlife/Livestock Interface: Implications for Wildlife, Livestock and Human Health, AHEAD (Animal Health for the Environment And Development), 14th and 15th September, 2003. Occasional Paper of the IUCN Species Survival Commission No. 30. 1–13 pp.

Kossila V. 1987. The availability of crop residues in developing countries in relation to livestock populations. In J.D Reed, B.S. Capper & P.J.H. Neate eds. 1988. *Plant breeding and the nutritive value of crop residues.* Proceedings of a workshop held at ILCA, Addis Ababa, Ethiopia. International Livestock Centre for Africa, Addis Ababa. (Available at www.ilri.cgiar.org/InfoServ/Webpub/Fulldocs/X5495e/x5495e03.htm).

Krapac, I.G., Dey, W.S., Roy, W.R., Smyth, C.A., Storment, E., Sargent, S.L. & Steele J.D. 2002. Impacts of swine manure pits on groundwater quality. *Environmental Pollution*, 120(2): 475–492.

Krystallis, A. & Arvanitoyannis, I.S. 2006. Investigating the concept of meat quality from the consumers perspective: the case of Greece. *Meat Science*, 72: 164–176.

Kumar, M.D. & Singh, O.P. 2001 Market instruments for demand management in the face of scarcity and overuse of water in Gujarat, Western India. *Water Policy,* 3: 387–403.

Lal, R. 1995. Erosion–crop productivity relationships for soils of Africa. *Soil Science Society of America Journal*, 59: 661–667.

Lal, R. 1997. Residue management conservation tillage and soil restoration for mitigating greenhouse effect by CO2-enrichment. *Soil and Tillage Research*, 43: 81–107.

Lal, R. 1998. Soil erosion impact on agronomic productivity and environment quality. *Critical Reviews in Plant Sciences*, 17(3): 19–464.

Lal, R., Kimble, J., Follett, R. & Cole, C.V. 1998 *Potential of US cropland for carbon sequestration and greenhouse effect mitigation.* Chelsea, Michigan, USA, Sleeping Bear Press. 128 pp.

Lal, R. & Bruce, J.P. 1999 The potential of world cropland soils to sequester C and mitigate the greenhouse effect. *Environmental Science and Policy,* 2: 177–185.

Lal, R. 2001. The potential of soils of the tropics to sequester carbon and mitigate the greenhouse effect. *Adv. Agron.*, 76: 1–30.

Lal, R. 2004a. Soil carbon sequestration impacts on global climate change and food security. *Science*, 304 (5677): 1623–1627.

Lal, R. 2004b. Carbon sequestration in dryland ecosystems. *Environmental Management* 33(4): 528–544.

Lambin, E.F., Turner, B.L., Geist, H.J., Agbola, S.B., Angelsen, A., Bruce, J.W., Coomes, O., Dirzo, R., Fischer, G., Folke, C., George, P.S., Homewood, K., Imbernon, J., Leemans, R., Li, X., Moran, E.F., Mortimore, M., Ramkrishnan, P.S., Richards, J.F., Skanes, H., Steffen, W.L., Stone, G.D., Svedin, U., Veldkamp, T.A., Vogel, C. & Zu, J. 2001. The causes of land use and land-cover change: moving beyond the myths. *Global Environmental Change*, 11: 261–269.

LandScan Project. Oak Ridge National Laboratory, (Available at http://www.ornl.gov/sci/landscan).

Larsen, R.E. 1995. Manure loading into streams from direct fecal deposits - Fact Sheet No. 25. Rangeland Watershed Program; UC Cooperative Extension and USDA Natural Resources Conservation Service; California Rangelands Research and Information Center, Agronomy and Range Science, Davis, USA, University of California.

Le Bel, S., Gaidet, N., Snoden, M., Le Doze, S. & Tendayi, N. 2004. *Communal game ranching in the mid-Zambezi valley: Challenges of local empowerment and sustainable game meat production for rural communities*. Proceedings de: La Faune Sauvage: Une Ressource Naturelle, 6ème symposium international sur l'utilisation de la faune sauvage. Paris France from 6–9 July 2004.

Le Quesne, T. & McNally, R. 2004. *The green buck: Using economic tools to deliver conservation goals*. WWF field guide. The WWF Sustainable Economics Network, WWF. pp. 69.

LEAD. 2002. *AWI Nutrient balance*, (Available at http://www.virtualcentre.org/en/dec/nutrientb/default.htm).

Lenné, J.M., Fernandez-Rivera, S. & Bümmel, M. 2003. Approaches to improve the utilization of food-feed crops – synthesis, *Field Crops Research*, 84(1-2): 213–222.

Leonard, D.K. 2006. *The political economy of international development and pro-poor livestock policies*. PPLPI Working Paper No. 35. FAO, Rome.

Lerner, J., Matthews, E. & Fung, I. 1988. Methane emissions from animals: a global high resolution database. *Global Biogeochemical Cycles*, 2, p. 139–156.

Leslie, R. (ed.), 1999. Coral reefs: Assessing the threat. In *World resources: a guide to the global environment 1998–99*, American Association for the Advancement of Science, p 193.

Lind, L., Sjögren, E., Melby, K. & Kaijser, B. 1996. DNA fingerprinting and serotyping of campylobacter jejuni isolates from epidemic outbreaks. *Journal of Clinical Microbiology*, 34(4): 892–896.

Lipper, L., Pingali, P.L. & Zurek, M. 2006. *Less-favoured areas: Looking beyond agriculture towards ecosystem services*. Agricultural and Development Economics Division (ESA) Working Paper, forthcoming. Rome, FAO.

Livestock In Development. 1999. *Livestock in poverty focused development.* Crewkerne: Livestock in development.

Loreau, M. & Oteng-Yeboah, A. 2006. Diversity without representation. *Nature*, 442: 245–246.

Lorimor, J., Fulhage, C., Zhang, R., Funk, T., Sheffield, R., Sheppard, D.C. & Newton, G.L. 2001. *Manure management strategies/technologies*. White paper summaries, National center for manure and waste management.

LPES. 2005. *Livestock and Poultry Environmental Stewardship Curriculum: A national educational program*. (Available at http://www.lpes.org/Lessons/Lesson01/1_Environmental_Stewardship.html).

Luke, G.J. 1987. *Consumption of water by livestock*. Resource Management Technical Report No. 60, Department of Agriculture Western Australia.

MacDonald, N.W., Randlett, D.L. & Zak, D.R. 1999. Soil warming and carbon loss from a Lake States Spodosol. *Soil Science Society of America Journal*, 63(1): 211–218.

Mack, R.N. 1989. Temperate grasslands vulnerable to plant invasions: Characteristics and consequences. Pages 155–179. In: Drake, J.A., H.A. Mooney, F. di Castri, R.H. Groves, F.J. Kruger, M. Rejmanek, and M. Williamson, eds., *Biological invasions: A global perspective*. John Wiley & Sons Ltd.

MAF–NZ. 2005. *Environmental consequences of removing agricultural subsidies*. New Zealand, Ministry of Agriculture and Forestry. (Available at http://www.maf.govt.nz/mafnet/rural-nz/sustainable-resource-use/resource-management/environmental-effects-of-removing-subsidies/agref004.htm, accessed on 22 July 2005).

MAFF-UK. 1998. *Ministry of Agriculture Fisheries and Food Code of agricultural practices for the protection of water.* London MAFF publication.

MAFF-UK. 1999. *Agricultural land sales and prices in England*. Ministry of Agriculture, Fisheries and Food, UK, (Available at http://statistics.defra.gov.uk/esg/pdf/alp9906.pdf).

Mainstone, C.P. & Parr, W. 2002. Phosphorus in rivers—ecology and management. *Science of the Total Environment*, 282: 25–47.

Margni, M., Jolliet, O., Rossier, D. & Crettaz, P. 2002. Life cycle impact assessment of pesticides on human health and ecosystems Agriculture. *Ecosystems & Environment*, 93(1–3): 379–392.

Marris, E. 2005. Conservation in Brazil: The forgotten ecosystem. *Nature*, 437: 944–945.

Marzocca, A. 1984. *Manuel de Malezas*, 3rd Edición. Editorial Hemisferio Sur, Buenos Aires, 580 pp.

Mather, A. 1990. *Global forest resources*. Portland, Oregon, USA. Timber Press.

Matson, P.A., Parton, W.J., Power, A.G. & Swift M.J. 1997. Agricultural intensification and ecosystem properties. *Science*, 277(5325): 504–509.

Matthews, E., Payne, R., Rohweder, M. & Murray, S. 2000. *Pilot analysis of global ecosystems: Forest ecosystems*. Research Report, World Resources Institute, Washington, DC, 100 pp.

May, P.H., Boyd, E., Veiga, F., & Chang, M. 2003. *Local sustainable development effects of forest carbon projects in Brazil and Bolivia: a view from the field*. Shell Foundation and IIED – International Institute for Environment and Development: Rio de Janeiro, October 2003. (Available at http://www.iied.org/).

Mayrand, K., Dionne S., Paquin, M., Ortega, G.A & Marron, L.F. 2003. *The economic and environmental impacts of agricultural subsidies: A look at Mexico and other OECD countries*. Montreal, Canada.

McCann, L. 2004. Induced institutional innovation and transaction costs: the case of the australian national native title tribunal. Review of Social Economy, Volume 62, No. 1, March 2004, Routledge, part of the Taylor & Francis Group. pp. 67–82(16).

Mcdonnell, M.J., Pickett, S.T.A., Groffman, P., Bohlen, P., Pouyat, R.V., Zipperer, W.C., Parmelee, R.W., Carreiro, M.M. & Medley, K. 1997. Ecosystem processes along an urban-to-rural gradient. *Urban Ecosystems*, 1(1): 21–36.

McDowell, R.W., Drewry, J.J., Paton, R.J., Carey, P.L., Monaghan, R.M. & Condron, L.M. 2003. Influence of soil treading on sediment and phosphorus losses in overland flow. *Australian Journal of Soil Research*, 41(5): 949–961.

McKergow, L.A., Weaver, D.M., Prosser, I.P., Grayson, R.B. & Reed A.E.G. 2003. Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, Western Australia. *Journal of Hydrology*, 270(3–4): 253–272.

McLeod, A., Morgan, N., Prakash, A. & Hinrichs, J. 2005. *Economic and social impacts of avian influenza*. Meeting on Avian Influenza – Geneva, 7–9 November 2005, FAO. (Available at http://www.fao.org/ag/againfo/subjects/en/health/diseases-cards/CD/documents/Economic-and-social-impacts-of-avian-influenza-Geneva.pdf).

Meat Research Corporation (MRC). 1995. *Identification of nutrient sources, reduction opportunities and treatment options for Australian abattoirs and rendering plants.* Project No. M.445. Prepared by Rust PPK Pty Ltd and Taylor Consulting Pty Ltd.

Médard, C. 1998. Dispositifs électoraux et violences ethniques: réflexions sur quelques stratégies territoriales du régime Kényan. *Politique Africaine*, 70: 32–39.

Melse, R.W. & van der Werf, A.W. 2005. Biofiltration for mitigation of methane emissions from animal husbandry. *Environmental Science & Technology*, 39(14): 5460–5468.

Melvin, R.G. 1995. *Non point Sources of Pollution on Rangeland* – Fact Sheet No. 3. Rangeland Watershed Program; U.C. Cooperative Extension and USDA Natural Resources Conservation Service; California Rangelands Research and Information Center – Agronomy and Range Science – UC Davis.

Melvin, R.G., Larsen, R.E., McDougald, N.K., Tate, K.W., Gerlach, J.D. & Fulgham, K.O. 2004. Cattle grazing has varying impacts on stream-channel erosion in oak woodlands. *California Agriculture*, 58(3): 138.

Mendis, M. & Openshaw, K. 2004. The clean development mechanism: making it operational. *Environment, Development and Sustainability*, 6(1-2): 183–211.

Mengjie, W. & Yi, D. 1996. The importance of work animals in rural China. *World Animal Review*, FAO. pp. 65–67.

Menzi H. & Kessler J. 1998. *Heavy metal content of manures in Switzerland*. Proceedings of the 8th international Conference on the FAO ESCORENA Network on Recycling of Agricultural, Municipal and Industrial Residues in Agriculture (Formerly Animal Waste Management). Rennes, France, 26–29 May 1998.

Menzi, H. 2001. *Needs and implications for good manure and nutrient management in intensive livestock production in developing countries*. Area Wide Integration Workshop, unpublished.

Mestbank. 2004. *Voortgang 2004 aangaande het mest beleid in Vlaanderen*, (Available at http://www.vlm.be/Mestbank/FAQ/algemeen/04voortgangsrapport.pdf).

Metting, F., Smith, J. & Amthor, J. 1999. Science needs and new technology for soil carbon sequestration. In: N. Rosenberg, R. Izaurralde & E. Malone, eds. Carbon sequestration in soils. Science monitoring and beyond, pp. 1–34. Proc. St. Michaels Workshop. Columbus, USA, Battelle Press.

Micheli, E.R. & Kirchner, J.W. 2002. Effects of wet meadow riparian vegetation on streambank erosion. 1. Remote sensing measurements of streambank migration and erodibility. *Earth Surface Processes and Landforms*, 27(6): 627–639.

Milchunas, D.G. & Lauenroth, W.K. 1993. A quantitative assessment of the effects of grazing on vegetation and soils over a global range of environments. *Ecological Monographs*, 63(4): 327–366.

MEA. 2005a. *Ecosystems and human well-being: synthesis*, Washington, DC, Island Press.

MEA. 2005b. *Ecosystems and human well-being: biodiversity synthesis,* Washington, DC, World Resources Institute.

Miller, J.J. 2001. *Impact of intensive livestock operations on water quality.* Proceedings of the Western Canadian Dairy Seminar.

Milne, J.A. 2005. Societal expectations of livestock farming in relation to environmental effects in Europe. *Livestock Production Science*, 96(1): 3–9.

Miner, J.R., Buckhouse, J.C. & Moore, J.A. 1995 Will a Water Trough Reduce the Amount of Time Hay-Fed Livestock Spend in the Stream (and therefore improve water quality)? Fact Sheet No. 20 Rangeland Watershed Program; UC Cooperative Extension and USDA Natural Resources Conservation Service; California Rangelands Research and Information Center, Agronomy and Range Science, Davis, USA, University of California.

Ministério da Ciência e Tecnologia. 2002. *Primeiro inventário brasileiro de emissões antrópicas de gases de efeito estufa: Emissões de metano da pecuária.* Empresa Brasileira de Pesquisa Agropecuária (EMBRAPA) – Ministério da Ciência e Tecnologia. (Available at www.ambiente.sp.gov.br/proaong/SiteCarbono/2/Pecuaria.pdf).

Ministry of Science & Technology. 2004. *Brazil's initial national communication to the united nations framework convention on climate change*. Ministry of Science and Technology, General Coordination on Global Climate Change. Brasilia, Brazil. p. 271.

Mittermeier, R.A., Robles-Gil, P., Hoffmann, M., Pilgrim, J.D., Brooks, T.B., Mittermeier, C.G., Lamoreux, J. L. & Fonseca, G.A.B. 2004. Hotspots Revisited: Earth's Biologically Richest and Most Endangered Ecoregions. Mexico City, CEMEX. 390 pp.

Mizutani, F., Muthiani, E, Kristjanson, P. & Recke, H. 2005. Impact and value of wildlife in pastoral livestock production systems in Kenya: Possibilities for healthy ecosystem conservation and livestock development for the poor. In *conservation and development interventions at the wildlife/livestock interface implications for wildlife, livestock and human health.* Edited and compiled by Steven A. Osofsky. Proceedings of the Southern and East African Experts Panel on Designing successful conservation and development interventions at the wildlife/livestock interface: implications for wildlife, livestock and human health, AHEAD (Animal Health for the Environment And Development), 14–15 September, 2003. Occasional Paper of the IUCN Species Survival Commission No. 30. pp. 121–132.

Molden, D. & de Fraiture, C. 2004. *Investing in water for food, ecosystems and livelihoods*. Comprehensive Assessment of Water Management in Agriculture, International Water Management Institute (IWMI).

Monteny, G.J., Bannink, A. & Chadwick, D. 2006. Greenhouse gas abatement strategies for animal husbandry. *Agriculture, Ecosystems and Environment*, 112:163–170.

Mooney, H.A. 2005. *Invasive alien species: the nature of the problem.* In: *Invasive alien species: a new synthesis*, p. 1–15, Mooney, H.A., Mack, R.N., McNeely, J.A., Neville, L.E., Schei, P.J. and Waage, J.K. eds. SCOPE 63, Washington, DC, Island Press.

Morrison, J.A., Balcombe, K., Bailey, A., Klonaris, S. & Rapsomanikis, G. 2003. Expenditure on different categories of meat in Greece: the influence of changing tastes. *Agricultural Economics*, 28: 139–150.

Morse and Jackson. 2003. Fate of a representative pharmaceutical in the environment. Final report submitted to Texas Water Resources Institute. Texas Tech University.

Mosier, A., Wassmann, R., Verchot, L., King, J. & Palm, C. 2004. Methane and nitrogen oxide fluxes in tropical agricultural soils: sources, sinks and mechanisms. *Environment, Development and Sustainability,* 6(1–2): 11–49

Mosley, J.C., Cook, P.S., Griffis, A.J. & O'Laughlin, J. 1997. *Guidelines for managing cattle grazing in riparian areas to protect water quality: Review of research and best management practices policy*. Report No. 15, Policy Analysis Group (PAG) Report Series. Idaho Forest, Wildlife and Range Policy Analysis Group.

Mosquera-Losada, M.R., Rigueiro-Rodríguez, A. & McAdam, J., eds. 2004. *Proceedings of an International Congress on Silvopastoralsim and sustainable management.* Lugo, Spain, April 2004. Walingford, UK, CABI Publishing.

Mott, J.J. 1986. Planned invasions of Australian tropical sanannas. In: Groves, R.H. and Burdon, J.J., eds., *Ecology of biological invasions*, pp. 89–96., Cambridge, UK, Cambridge University Press.

Muirhead, R.W., Davies-Colley, R.J., Donnison, A.M. & Nagels, J.W. 2004. Faecal bacteria yields in artificial flood events: quantifying in-stream stores. *Water Research*, 38(5): 1215–1224.

Muller, W. & Schneider, B. 1985. *Heat, water vapour and CO2 production in dairy cattle and pig housing*. Part 1. Provisional planning data for the use of heat exchangers and heat pumps in livestock housing. (IN GERMAN) Tieraztliche Umschau 40, 274–280.

Mulongoy, K.J. & Chape, S.P., eds. 2004. *Protected areas and biodiversity: An overview of key issues*. UNEP-WCMC Biodiversity Series 21. The Secretariat of the Convention on Biological Diversity(CBD) and the UNEP World Conservation Monitoring Centre (UNEP-WCMC), Cambridge, UK.

Murgueitio, E. 2004. *Silvopastoral systems in the Neotropics*. Proceedings of an international congress on Silvopastoralism and Sustainable Management, Lugo, Spain, April 2004.

Mwendera, E.J. & Mohamed Saleem, M.A. 1997. Hydrologic response to cattle grazing in the Ethiopian highlands. *Agriculture, Ecosystems & Environment*, 64(1): 33–41.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., de Fonseca, G.A.B. & Kent, J. 2000. Biodiversity hotspots for conservation priorities. *Nature*, 403: 853–858.

Myers, R.J.K. & Robbins, G.B. 1991. Sustaining productive pastures in the tropics 5: maintaining productive sown grass pastures. *Tropical Grasslands*, 25: 104–110.

Myers, T.J. & Swanson, S. 1995. Long-term aquatic habitat restoration: Mahogany Creek, Nevada, as a case study. *Water Resources Bulletin*, 32(2): 241–252.

Nagle, G.N. & Clifton, C.F. 2003. Channel changes over 12 years on grazed and ungrazed reaches of Wickiup Creek in eastern Oregon. *Physical Geography*, 24(1): 77–95.

Napier, T. 2000. Soil and water conservation policy approaches in North America, Europe, and Australia. *Water Policy*, 1(6): 551–565.

NASA. 2005. *Global temperature trends: 2005 summation*, (Available at http://data.giss.nasa.gov/gistemp/2005/).

National Conservation Buffer Team. 2003. Leaflet.

National Park Service. 2004. US Department of the Interior. *Geologic resource monitoring parameters: Soil and sediment erosion*. (Available at http://www2.nature.nps.gov/geology/monitoring/soil_erosion.pdf).

National Public Lands Grazing Campaign. 2004. *Livestock and water.* (Available at http://www.publiclandsranching.org/htmlres/fs_cows_v_water.htm Consulted in 2004.

National Research Council. 1981. *Effects of environment on nutrient requirements of domestic animals*. Subcommittee on Environmental Stress, Committee on Animal Nutrition, National Research Council, Washington DC, National Academy Press. pp. 168.

National Research Council. 1985. *Nutrient requirements of sheep* – Sixth Revised Edition, Subcommittee on Sheep Nutrition, Committee on Animal Nutrition, National Research Council. Washington, DC, National Academy Press. 112 pp.

National Research Council. 1987. *Predicting feed intake of food-producing animals.* Subcommittee on Feed Intake, Committee on Animal Nutrition National Research Council, Washington, DC, National Academy Press. 248 pp.

National Research Council. 1993. *Managing global genetic resources: Livestock.* Committee on Managing Global Genetic Resources: Agricultural Imperatives. Washington, DC, National Academy Press.

National Research Council. 1994. *Nutrient requirements of poultry* – Ninth Revised Edition. Subcommittee on Poultry Nutrition, Committee on Animal Nutrition, National Research Council, Washington, DC, National Academy Press. 176 pp.

National Research Council. 1998. *Nutrient requirements of swine* – 10th Revised Edition. Subcommittee on Swine Nutrition, Committee on Animal Nutrition, National Research Council, Washington, DC, National Academy Press. 210 pp.

National Research Council. 2000a. *Nutrient requirements of beef cattle* – Seventh Revised Edition. Subcommittee on Beef Cattle Nutrition, National Research Council, Washington, DC, National Academy Press. 248 pp.

National Research Council. 2000b. *Clean coastal waters: Understanding and reducing the effects of nutrient pollution.* Washington, DC, National Academy Press.

National Research Council. 2003. *Air emissions from animal feeding operations: current knowledge, future needs.* Ad Hoc Committee on Air Emissions from Animal Feeding Operations, Committee on Animal Nutrition, National Research Council, Washington, DC, National Academy Press. 263 pp.

NatureServe. 2005. *NatureServe Explorer: An online encyclopedia of life* [web application]. Version 4.5. NatureServe, Arlington, Virginia.(Available at www.natureserve.org/explorer. Accessed July 2005).

Naylor, R., Steinfeld, H., Falcon, W., Galloway, J., Smil, V., Bradford, E., Alder, J. & Mooney, H. 2005 Losing the links between livestock and land. *Science*, 310: 1621–1622.

Nelson, P.N., Cotsaris, E. and Oades, J.M. 1996. Nitrogen, phosphorus,and organic carbon in streams draining two grazed catchments. *Journal of Environmental Quality*, 25 (6:1221–1229.

Neumann, C.G., Bwibo, N.O., Murphy, S.P., Sigman, M., Whaley, S., Allen, L.H., Guthrie, D., Weiss, R.E. & Demment, M.W. 2003. Animal Source Foods Improve Dietary Quality, Micronutrient Status, Growth and Cognitive Function in Kenyan School Children: Background, Study Design and Baseline Findings. The American Society for Nutritional Sciences J. Nutr. 133:3941S–3949S.

Ni, J-Q., Hendriks, J., Coenegrachts, J. & Vinckier, C. 1999. Production of carbon dioxide in a fattening pig house under field conditions. 1. Exhalation by pigs. *Atmospheric Environment*, 33: 3691–3696.

Nicholson, F.A., Smith, S.R., Alloway, B.J., Carlton-Smith, C. & Chambers B.J. 2003. An inventory of heavy metals inputs to agricultural soils in England and Wales. *The Science of the Total Environment*, 311: 205–219.

Nielsen, L.H. & Hjort-Gregersen, K. 2005. Greenhouse gas emission reduction via centralized biogas co-digestion plants in Denmark. *Agric. Ecosys. Environ*. 112.

Niklinska, M., Maryanski, M. & Laskowski, R. 1999. Effect of temperature on humus respiration rate and nitrogen mineralization: Implications for global climate change. *Biogeochemistry*, 44: 239–257.

Nill, K. 2005. *US soybean production is more sustainable than ever before*. Press Release, September 2005. American Soybean Association.

NOAA. 2006. *Trends in atmospheric carbon dioxide*. NOAA/Earth System Research Laboratory, Global Monitoring Division.

Nori, M., Switzer, J. & Crawford, A. 2005. *Herding on the brink: Towards a global survey of pastoral communities and conflict*. International Institute for Sustainable Development.

Norton, R.D. 2003. Agricultural development policy: Concepts and experiences. Food and Agriculture Organization of the United Nations. UK, John Wiley & Sons Ltd.

Norton-Griffiths, M. 1995. Economic incentives to develop the rangelands of the Serengeti: Implications for wildlife conservation. In *Serengeti II: research, management and conservation of an ecosystem*. A.R.E. Sinclair and P. Arcese, eds., Chicago, USA, University of Chicago Press. 14 pp.

Notermans, S., Dufrenne, J. & Oosterom, J. 1981. Persistence of Clostridium Botulinum type B on a cattle farm after an outbreak of Botulism. *Applied and Environmental Microbiology*, 41(1): 179–183.

Novotny, V., Imhoff, K.R., Olthof, M. & Krenkel, P.A. 1989. *Handbook of urban drainage and wastewater*. New York, USA, Wiley & Sons Publishers.

Nye, P.H. & Greenland, D.J. 1964. Changes in the soil after clearing tropical forest, *Plant and Soil*, 21(1): 101–112.

Odling-Smee, L. 2005. Dollars and sense. *Nature*, 437: 614–616.

OECD. 1999. *Agricultural water pricing in OECD Countries*. Paris, Organization for Economic Cooperation and Development. 1999.

OECD. 2001. *Towards more liberal agricultural trade*. Policy Brief. Paris: Organization for Economic Co-operation and Development.

OECD. 2002. *Farm household income issues in OECD countries: A synthesis report.* Paris. Organization for Economic Co-operation and Development.

OECD. 2004. *Agriculture and the environment: lessons learned from a decade of OECD work.* Paris. June 2004.

OECD. 2006. *Agricultural policies in OECD countries – at a glance.* Paris. 2006.

Oldeman, L.R. 1994. The global extent of land degradation. In Greenland, D.J. & I. Szabolcs, eds., *Land resilience and sustainable land use*, 99–118. Wallingford, UK, CABI Publishers.

Oldeman, L.R. & Van Lyden, G.W.J. 1998. Revisiting the GLASOD methodology. In R. Lal, W.H. Blum, C. Valentine & B.A. Stewart, eds. *Methods for assessment of soil degradation,* pp. 423–440. Boca Raton, USA, CRC/Lewis Publishers.

Olea, L., Lopez-Bellido, R.J. & Poblaciones, M.J. 2004. *European types of silvopastoral systems in the Mediterranean area: dehesa*. Proceedings of an international congress on Silvopastoralism and Sustainable Management, Lugo, Spain, April 2004.

Olivier, J.G.J., Brandes, L.J., Peters, J.A.H.W. & Coenen, P.W.H.G. 2002. Greenhouse gas emissions in the Netherlands 1990–2000. National Inventory Report 2002. Rijksinstituut voor Volkagezondheld en Milieu. RIVM Rapport 773201006. pp. 150.

Olson, D.M. & Dinerstein, E. 1998. The Global 200: A representation approach to conserving the earth's most biologically valuable ecoregions. *Conservation Biology*, 12: 502–515.

Olson, D. M., Dinerstein, E. 2002. The Global 200: Priority ecoregions for global conservation. Annals of the Missouri Botanical Garden 89 : 125–126.

Olson, M.E., O'Handley, R.M., Ralston, B.J., McAllister, T.A. & Thompson, R.C.A. 2004. Update on Cryptosporidium and Giardia infections in cattle. *Trends in Parasitology*, 20(4): 185–191.

Ong, C., Moorehead, W., Ross, A. & Isaac-Renton, J. 1996. Studies of Giardia spp. and Cryptosporidium spp. in two adjacent watersheds. *Applied and Environmental Microbiology*, 62(8): 2798–2805.

Ongley, E.D. 1996. *Control of water pollution from agriculture*. FAO Irrigation and Drainage Paper No. 55, FAO, Rome.

Orr, J.C., Fabry, V.J., Aumont, O., Bopp, L., Doney, S.C., Feely, R.A., Gnanadesikan, A., Gruber, N., Ishida, A., Joos, F., Key, R.M., Lindsay, K., Maier-Reimer, E., Matear, R., Monfray, P., Mouchet, A., Najjar, R.G., Gian-Kasper Plattner, Rodgers, K.B., Sabine, C.L., Sarmiento, J.L., Schlitzer, R., Slater, R.D., Totterdell, I.J., Weirig, Marie-France, Yamanaka, Y. & Yoo, A. 2005. Anthropogenic ocean acidification over the twentyfirst century and its impact on calcifying organisms. *Nature*, 437: 681–686.

Osoro, K., Celaya, R., Martinez, A. & Vasallo, J.M. 1999. Development of sustainable systems in marginal heathland regions. LSIRD Network Newsletter Issue 6. European Network for Livestock Systems and Integrated Rural Development.

Osterberg, D. & Wallinga, D. 2004. *Determinants of rural health*. American Journal of Public Health 94(10).

Ostermann, O.P. 1998. The need for management of nature conservation sites designated under Natura 2000. *Journal of Applied Ecology*, 35(6), 968–973.

Oweis, T.Y. & Hachum, A.Y. 2003. Improving water productivity in the dry areas of West Asia and North Africa. Chapter 11, pp 179–198, In J.W. Kijne, R. Barker, & D. Molden. 2003. *Water productivity in agriculture: Limits and opportunities for improvement.* Wallingford, UK, CABI Publishing.

Pagiola, S., Agostini, P., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., Ramirez, E., Rosales, M. & Pablo Ruiz, J. 2004. *Paying for biodiversity conservation services in agricultural landscapes*. Environment Department Paper No.96. Washington, DC, World Bank Environment Department, World Bank.

Pagiola, S., von Ritter, K. & Bishop, J. 2004. *Assessing the economic value of ecosystem conservation*. Environment Department Paper No.101. Washington, DC, World Bank Environment Department, World Bank.

Pallas, Ph. 1986. *Water for animals*. Land and Water Development Division, FAO. (Available at www.fao.org/docrep/R7488E/R7488E00.htm).

Parris, K. 2002. *Environmental impacts in the agricultural sector: using indicators as a tool for policy purposes*. Paper presented to the Commission for Environmental Cooperation Meeting: *Assessing the Environmental Effects of Trade*, Montreal, Canada, 17–18 January 2002.

Parry, M.L., Rosenzweig, C., Iglesias, A., Livermore, M. & Fischer, G. 2004. Effects of climate change on global food production under SRES emissions and socio-economic scenarios. *Global Environmental Change*, 14: 53–67.

Patten, D.T., Ohmart, R.D., Meyerhoff, R., Ricci, E. Shirley, D. Minckley W.L. & Kubly D.M. 1995. The Arizona Comparative Environmental Risk Project: Ecosystems – *Riparian ecosystems*, Section 2, Chapter 1. The Arizona Comparative Environmental Risk Project (ACERP) Report. Arizona EarthVision.

Patterson, B., Kasiki, S., Selempo, E. & Kays, R. 2004. Livestock predation by lions (*Panthera leo*) and other carnivores on ranches neighboring Tsavo National Parks, Kenya. *Biological Conservation*, 119: 507–516.

Pauly, D. & Watson, R. 2003. Counting the last fish. *Scientific American Magazine*, 289(1): 34–39.

Pauly, D., Alder, J., Bennett, E., Christensen, V., Tyedmers, P. & Watson, R. 2003. The Futures for Fisheries. *Science*, 302(5649):1359–1361.

Paustian, K., Andren, O., Janzen, H.H., Lal, R., Smith, P., Tian, G., Tiessen, H., Van Noordwijk, M. & Woomer P.L. 1997. Agricultural soils as a sink to mitigate carbon dioxide emissions. *Soil Use and Management*, 13(4): 230–244.

Pearce, D. 2002. Environmentally Harmful Subsidies: Barriers to Sustainable Development, Paper presented at the OECD workshop on environmentally harmful subsidies, Paris, 7–8 November 2002.

Perrings, C. & Touza-Montero, J. 2004. *Spatial interactions and forests management: policy issues. In proceedings of the Conference on Policy Instruments for Safeguarding Forest Biodiversity – Legal and Economic Viewpoints.* The Fifth International BIOECON Conference 15th–16th January 2004, House of Estates, Helsinki. (Available at http://www.metla.fi/julkaisut/workingpapers/2004/mwp001-03.pdf).

Perry, C.J., Rock, M. & Seckler, D. 1997. *Water as an economic good: a solution or a problem?* In: *water: economics, management and demand,* eds. M. Kay, T. Franks, L.E. Smith & F.N. Spon, pp. 3–10.

Phoenix, G.K., Hicks, W.K., Cinderby, S., Kuylenstierna, J.C.I., Stock, W.D., Dentener, F.J., Giller, K.E., Austin, A.T., Lefroy, R.D.B., Gimeno, B.S., Ashmore, M.R. & Ineson, P. 2006. Atmospheric nitrogen deposition in world biodiversity hotspots: The need for a greater global perspective in assessing N deposition impacts. *Global Change Biology*, 12(3): 470–476.

Pidwirny, M. 1999. *Fundamentals of physical geography: Introduction to the hydrosphere* (Chapter 8). Department of Geography, Okanagan University College.

Pierre, C. 1983. US agriculture and the environment. *Food Policy*, 8(2): 99–110.

Pieters, J. 2002. *When removing subsidies benefits the environment: Developing a checklist based on the conditionality of subsidies*. Paper presented at the OECD workshop on environmentally harmful subsidies, 7–8 November, Paris, France.

Pimentel, D., Berger, B., Filiberto, D., Newton, M., Wolfe, B., Karabinakis, E., Clark, S., Poon, E., Abbett, E. & Nandagopal, S. 2004. *Water resources, agriculture and the environment*.

Pingali, P.L. & Heisey P.W. 1999. *Cereal crop productivity in developing countries: past trends and future prospects.* CIMMYT Working Paper 99–03. International Maize and Wheat Improvement Centre (CIMMYT), (Available at www.cimmyt.org/Research/Economics/map/research_results/working_papers/pdf/EWP 2099_03.pdf).

Pons, A., Couteau, M., de Beaulieu, J.L. & Reille, M. 1989. The plant invasions in Southern Europe from the paleoecological point of view. In di Castri, F., Hansen, A.J., Debussche, M. (Eds.), *Biological invasions in Europe and the Mediterranean Basin*, Kluwer Academic Publications, Dordrecht.

Popkins, B., Horton, S. & Kim, S. 2001. The nutrition transition and prevention of diet-related chronic diseases in Asia and the Pacific. *Food and Nutrition Bulletin*, 22 (4: Suppl.). Tokyo, United Nations University Press.

Porter, G. 2003. *Agricultural trade liberalisation and the environment in North America: Analysing the "production effect"*. Prepared for "the second North American Symposium on Assessing the Environmental Effects of Trade". Commission for Environmental Cooperation.

Postel, S. 1996. *Dividing the waters: Food security, ecosystem health, and the new politics of scarcity*. Worldwatch Paper 132, September 1996. 76 pp.

Pott, R. 1998. Effects of human interference on the landscape with special reference to the role of livestock. In M.F. WallisDeVries, J.P. Bakker, & S.E. Van Wieren, eds., *Grazing and conservation management*. Kluwer, Dordrecht, pp. 107–134.

Pretty, J.N., Brett, C., Gee, D., Hine, R.E., Mason, C.F., Morison, J.I.L., Raven, H., Rayment, M.D. & van der Bijl, G. 2000. An assessment of the total external costs of UK agriculture. *Agricultural Systems*, 65(2): 113–136.

Price, L., Worrell, E., Martin, N., Lehman, B. & Sinton, J. 2000. *China's industrial sector in an international context*. Environmental Energy Technologies Division. Lawrence Berkeley National Laboratory. Report LBNL 46273. p. 17. (Available at http://eetd.lbl.gov/ea/IES/iespubs/46273.pdf).

Prince, S. D. & S. N. Goward. 1995. Global primary production: a remote sensing approach. Journal of Biogeography 22: 815–835.

Purcell, D.L. & Anderson, J.R. 1997. *Agricultural Research and Extension: Achievements and problems in national systems*; World Bank Operations Evaluation Study, Washington DC, World Bank.

Purdue University. 2006. *Department of Agronomy: Crop, Soil and Environmental Sciences.* (accessed April 2006. Available at http://www.agry.purdue.edu/index.asp).

Quinlan Consulting. 2005. *The effects of deforestation on the Hydrological Regime.* (Available at: http://www.headwaterstreams.com/hydrology.html, accessed April 2006).

Quist, & Chapela, I. 2001. Transgenic DNA introgressed into traditional landraces in Oaxaca, Mexico. *Nature*, 414(6863): 541–543.

Rabalais, N.N., Turner, R.& Scavia, D. 2002. Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi river. *BioScience*, 52(2), 129–142.

Rae, A. 1998. The effects of expenditure growth and urbanisation on food consumption in East Asia: a note on animal products, *Agricultural Economics*, 18(3): 291–299.

Rae, A. & Strutt, A. 2003. Agricultural trade reform and environmental pollution from livestock in OECD countries; Paper presented to the sixth Annual Conference on Global Economic Analysis, The Hague, 12–14 June 2003.

RAMSAR. 2005. The Ramsar Convention on wetlands: Background papers on wetland values and functions, (Available at http://www.ramsar.org/info/values_intro_e.htm).

Ranjhan, S.K. 1998. *Nutrient Requirements of livestock and Poultry.* New Delhi, Indian council of Agricultural Research, India.

Rao, J.M. 1989. Taxing agriculture: instruments and incidence; *World Development,* 17(6), 1989: 813.

Reddy, K.R., Kadlec, R.H., Flaig, E. & Gale, P.M. 1999. Phosphorus retention in streams and wetlands: a review. *Critical Reviews in Environmental Science and Technology*, 29(1): 83–146.

Redecker, B., Hardtle, W., Finck, P., Riecken, U. & Schroder, E., eds. 2002. Pasture, landscape and nature conservation. Springer. 435 pp.

Redmon, L.A. 1999. *Conservation of soil resources on lands used for grazing*. Proceedings of the third Workshop, conservation and use of natural resources and marketing of beef cattle, 27–29 January 1999. Monterrey, Mexico.

Reich, P.B., Peterson, D.A., Wrage, K. & Wedin, D. 2001. Fire and vegetation effects on productivity and nitrogen cycling across a forest-grassland continuum. *Ecology*, 82: 1703–1719.

Reich, P.F., Numbem, S.T., Almaraz, R.A. & Eswaran, H. 1999. Land resource stresses and desertification in Africa. In E.M. Bridges, I.D. Hannam, L.R. Oldeman, F.W.T. Pening de Vries, S.J. Scherr & S. Sompatpanit, eds., *Responses to land degradation. Proceedings of the second international conference on land degradation and desertification*, Khon Kaen, Thailand. New Delhi, Oxford Press.

Reid, R., Thornton, P.K., Mccrabb, G., Kruska, R., Atieno, F. & Jones, P. 2004. Is it possible to mitigate greenhouse gas emissions in pastoral ecosystems of the tropics? *Environment, Development and Sustainability*, 6: 91–109.

Rejmánek, M., Richardson, D.M., Higgins, S.I., Pitcairn, M.J. & Grotkopp, E. 2005. Ecology of invasive plants: State of the art. In: *Invasive alien species, a new synthesis*, ed. H.A. Mooney, *et al.*, SCOPE 63, Island Press. 368 pp.

Relyea, R.A. 2004. The growth and survival of five amphibian species exposed to combinations of pesticides. Environmental Toxicology and Chemistry, 23:1737–1742

Renner, M. 2002. The anatomy of resource wars. Worldwatch Paper No. 162. Worldwatch Institute.

Renter, D.G., Sargeant, J.M., Oberst, R.D. & Samadpour, M. 2003. Diversity, frequency, and persistence of Escherichia coli O157 strains from range cattle environments. *Applied Environmental Microbiology*, 69(1): 542–547.

Requier-Desjardins, M. & Bied-Charreton, M. 2006. *Evaluation des coûts économiques et sociaux de la dégradation des terres et de la désertification en Afrique.* Centre d'économie et d'éthique pour l'environnement et le développement; Versailles, France, Université de Versailles Saint Quentin-en-Yvelines.

Rice, C.W. 1999. *Subcommittee on production and price competitiveness hearing on carbon cycle research and agriculture's role in reducing climate change*.

Richards, K. 2004. A brief overview of carbon sequestration economics and policy. *Environmental Management* 33(4): 545–558.

Ricketts, T.H., Dinerstein, E., Boucher. T., Brooks, T.M., Butchart, S. H. M., Hoffmann, M., Lamoreux, J.F., Morrison, J., Parr, M., Pilgrim, J.D., Rodrigues, A. S. L., Sechrest, W., Wallace, G. E, Berlin, K., Bielby, J., Burgess, N.D., Church, D.R., Cox, N., Knox, D., Loucks, C., Luck, G.W., Master, L.L, Moore, R., Naidoo, R., Ridgely, R., Schatz, G.E., Shire, G., Strand, H., Wettengel, W. & Wikramanayake, E. 2005. Pinpointing and preventing imminent extinctions. *Proceedings of the National Academy of Sciences* (PNAS), 102(51): 18497–18501.

Rihani, N. 2005. Cours supérieur de production animale. Instituto Agronómico Mediterráneo de Zaragoza, Spain.

Risse, L.M., Cabrera, M.L., Franzluebbers, A.J., Gaskin, J.W., Gilley, J.E., Killorn, R., Radcliffe, D.E., Tollner, W.E. & Zhang, H. 2001. Land application of manure for beneficial reuse. pp. 283–316. In: *Animal agriculture and the environment: national center for manure and animal waste management white papers.* J.M. Rice, D.F. Caldwell, F.J. Humenik, eds. St. Joseph, Michigan, USA, published by the American Society of Agricultural and Biological Engineers.

Ritter, W.F. & Chirnside, A.E.M. 1987. Influence of agricultural practices on nitrates in the water table aquifer. *Biological Wastes*, 19(3): 165–178.

Rodary, E. & Castellanet C. 2003. Les trois temps de la conservation. In E. Rodary, C. Castellanet & G. Rossi, eds. *Conservation de la nature et développement, l'intégration possible?* Paris. Karthala and GRET, 225–237.

Rogers, P., Bhatia, R. & Huber, A. 1998. *Water as a social and economic good: How to put the principle into practice*. Global Water Partnership/Swedish International Development Cooperation Agency.

Rook, A.J., Dumont, B., Osoro, K., WallisDeFries, M.F., Parente, G. & Mills, J. 2004. Matching type of livestock to desired biodiversity outcomes in pastures – a review. Biological Conservation, 119:137–150.

Roost, N., Molden, D., Zhu, Z. & Loeve, R. 2003. *Identifying water saving opportunities: examples from three irrigation districts in China's Yellow River and Yangtze Basins.* Paper presented at the 1st International Yellow River Forum on River Basin Management Held in Zhengzhou, China, 12–15 May 2003

Rosales, M. & Livinets, S. 2005. Grazing and land degradation in CIS countries and Mongolia. In: Proceedings of the electronic conference "Grazing and land degradation in CIS countries and Mongolia". 10 June – 30 July 2005. (Available at http://www.lead.virtualcentre.org/ru/ele/econf_01_grazing/download/intro_en.pdf).

Rosegrant, M.W. & Binswanger, H. 1994. Markets in tradable water rights: Potential for efficiency gains in developing country irrigation. *World Development*, 22: 1613–1625.

Rosegrant, M.W., Leach, N. & Gerpacio, R.V. 1999. Meat or wheat for the next millennium? Alternative futures for world cereal and meat consumption. *Proceedings of the Nutrition Society*, 58: 219–234.

Rosegrant, M.W., Cai, X. & Cline, S.A. 2002. *Global water outlook to 2025, Averting an impending crisis. A 2020 vision for food, agriculture, and the environment initiative*. International food policy research institute (IFPRI) and International water management institute (IWMI).

Roth, E. 2001. *Water pricing in the EU: A review*. The European Environmental Bureau (EEB).

Rotz, C.A. 2004. Management to Reduce Nitrogen Losses in Animal Production. *Journal of Animal Science*, 82 (e. SUPPL.):E119–E137.

Roulet, P.A. 2004. Chasseur blanc, cœur noir? La chasse sportive en Afrique centrale. Une analyse de son rôle dans la conservation de la faune sauvage et le développement rural au travers des programmes de gestion de la chasse communautaire. Thèse de doctorat de Géographie. Laboratoire ERMES/IRD. Université d'Orléans, 2004.

Rudel, T.K. 1998. Is there a forest transition? Deforestation, reforestation and development. *Rural Sociology,* 64(4): 533–551.

Rudel, T.K., Bates, D. & Machinguiashi, R. 2002. A tropical forest transition? Agricultural changes, out-migration and secondary forests in the Ecuadorian Amazon. *Annals of the Association of American Geographers*, 92: 87–102.

Russelle, M.P. & Birr, A.S. 2004. Large-scale assessment of symbiotic dinitrogen fixation by crops: Soybean and alfalfa in the Mississippi river basin. *Agronomy Journal*, 96: 1754–1760.

Rutherford, J.C. & Nguyen, M.L. 2004. Nitrate removal in riparian wetlands: interactions between surface flow and soils. *Journal of Environmental Quality*, 33(3):1133-11–43.

Ruttan, V.W. 2001. *Technology, growth, and development: An induced innovation perspective*. New York, New York, USA: Oxford University Press.

Ryan, B. & Tiffany, D.G 1998. Energy use in Minnesota agriculture. *Minnesota Agricultural Economist* No. 693 Fall 1998, Minnesota Extension Service, University of Minnesota. (Available at www.extension.umn.edu/newsletters/ageconomist/components/ag237-693b.html).

Sainz, R. 2003. *Framework for calculating fossil fuel use in livestock systems*. Livestock, Environment and Development initiative report. (Available at ftp://ftp.fao.org/docrep/nonfao/LEAD/X6100E/X6100E00.PDF).

Salmon Nation. 2004. *Ten ways ranchers can help restore clean water and salmon*. (Available at http://www.4sos.org/howhelp/ranchers2.html).

Salter, L. 2004. *A clean energy future? The role of the CDM in promoting renewable energy in developing countries*. WWF International. pp. 11. (Available at http://www.panda.org/downloads/climate_change/liamsalterfullpapercorrected.pdf).

Sanderson, J., Alger, K., da Fonseca, G., Galindo-Leal, C., Inchausty, V.H. & Morrison, K. 2003. Biodiversity conservation corridors: Planning, implementing, and monitoring sustainable landscapes. Center for Applied Biodiversity Science- Conservation International. pp.43.

Saunders, C.S., Cagatay, S. & Moxey, A.P. 2004. *Trade and the environment: economic and environmental impacts of global dairy trade liberalisation*. Agribusiness and Economics Research Unit (AERU), Research report No. 267, February 2004.

Sauvé Jilene L., Goddard, T.W. & Cannon, K.R. 2000. *A preliminary assessment of carbon dioxide emissions from agricultural soils*. Paper presented at the Alberta Soil Science Workshop, February 22–24 2000, Medicine Hat, Alberta. (Available at http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/aesa8419?opendocument).

Schepers, J.S., Hackes, B.L. & Francis, D.D. 1982. Chemical water quality of runoff from grazing land in Nebraska: II contributing factors. *Journal of Environmental Quality*, 11(3): 355–359.

Scherf, B. ed. 2000. World watch list for domestic animal diversity, third edition. Rome. FAO/UNDP.

Scherr, S.J. & Yadav, S. 1996. *Land degradation in the developing world issues and policy options for 2020*. Washington, DC, IFPRI.

Schiere, H. & van der Hoek, R. 2000. *Livestock keeping in urban areas a review of traditional technologies*. FAO report.

Schmidhuber, J. & Shetty, P. 2005. The nutrition transition to 2030. Why developing countries are likely to bear the major burden. *Acta Agriculturae Scandinavica*, Section C Economy, 2(3–4): 150–166.

Schnittker, J. 1997. *The history, trade, and environmental consequences of soybean production in the United States*. Report to the World Wide Fund for Nature. 110 pp.

Schofield, K., Seager, J. & Merriman, R.P. 1990. The impact of intensive dairy farming activities on river quality: The eastern Cleddau catchment study. *Journal of the Institution of Water and Environmental Management*, 4(2): 176–186.

Scholes, M. & Andreae, M.O. 2000. Biogenic and pyrogenic emissions from Africa and their impact on the global atmosphere. *Ambio*, 29: 23–29.

Scholes, R.J., Schulze, E.D., Pitelka, L.F. & Hall, D.O. 1999. Biogeochemistry of terrestrial ecosystems. In: *The Terrestrial Biosphere and Global Change: Implications for Natural and Managed Ecosystems*. B. Walker, W. Steffen, J. Canadell & J. Ingram, eds., Cambridge, UK, University Press Cambridge, 271–303.

Schultheiß, U., Döhler H., Eckel, H, Früchtenicht, K., Goldbach, H., Kühnen, V., Roth, U., Steffens, G., Uihlein, A. & Wilcke, W. 2003. *Heavy metal balances in livestock farming.* Proceeding form the workshop: AROMIS - Assessment and reduction of heavy metal inputs into agro-ecosystems. 24–25 November 2003, Kloster Banz, Germany.

Schultz, R.C., Isenhart, T.M. & Colletti, J.P. 1994. *Riparian buffer systems in crop and rangelands*. Agroforestry and Sustainable Systems: Symposium Proceedings, August 1994.

Schulze, D.E. & Freibauer, A. 2005. Carbon unlocked from soils. *Nature*, 437: 205–206.

Schwartz, P. & Randall, D. 2003. *An abrupt climate change scenario and its implications for United States national security*, (Available at http://www.greenpeace.org/raw/content/international/press/reports/an-abrupt-climate-change-scena.pdf).

SCOPE 21. 1982. *The major biogeochemical cycles and their interactions*. Scientific Committee On Problems of the Environment (SCOPE). (Available at http://www.icsu-scope.org/downloadpubs/scope21/).

Scrimgeour, G.J. & S. Kendall. 2002. Consequences of livestock grazing on water quality and benthic algal biomass in a Canadian natural grassland plateau. *Environmental Management*, 29(6): 824–844.

Secretariat of the Convention on Biological Diversity. 2003. *Interlinkages between biological diversity and climate change*. Advice on the integration of biodiversity considerations into the implementation of the United Nations Framework Convention on Climate Change and its Kyoto protocol. Montreal, SCBD, 154 p. (CBD Technical Series No. 10).

Sharpley, A., Meisinger, J.J. Breeuwsma, A., Sims, J.T., Daniel, T.C. & Schepers, J.S. 1998. Impacts of animal manure management on ground and surface water quality. Pages 173–242 In J.L. Hatfield, & B.A. Stewart, eds., *Animal waste utilization: effective use of manure as a soil resource*. Chelsea, Michigan, USA, Ann Arbor Press.

Sharrow, S.H. 2003. Soil compaction during forest grazing. *The Grazier*, 317:2 Oregon State University.

Sheffer, M., Carpenter, S., Foley, J.A., Folke, C. & Walker, B. 2001. Catastrophic shifts in ecosystems. *Nature*, 413: 591–596.

Sheldrick, W., Syers, J.K. & Lingard, J. 2003. Contribution of livestock excreta to nutrient balances. *Nutrient Cycling in Agroecosystems*, 66(2): 119–131.

Shepherd, C.J., Pike, I.H. & Barlow, S.M. 2005. Sustainable feed resources of marine origin. European Aquaculture Society Special Publication 35: 59–66.

Shere, J.A., Bartlett, K.J. & Kaspar, W. 1998. Longitudinal study of Escherichia coli O157:H7 dissemination on four dairy farms in Wisconsin. *Applied and Environmental Microbiology*, 64(4): 1390–1399.

Shere, J.A., Kaspar, C.W., Bartlett, K.J., Linden, S.E., Norell, B., Francey, S. & Schaefer, D.M. 2002. Shedding of *Escherichia coli* O157:H7 in dairy cattle housed in a confined environment following waterborne inoculation. *Applied and Environmental Microbiology*, 68(4): 1947–1954.

Siebers, J., Binner, R. & Wittich, K.P. 2003. Investigation on downwind short-range transport of pesticides after application in agricultural crops. *Chemosphere*, 51(5): 397–407.

Siebert, S. Döll, P. & Hoogeveen, J. 2001, Global map of irrigated areas version 2.0, Center for Environmental Systems Research, Univerity of Kassel, Germany/ Food and Agriculture Organization of the United Nations, Rome.

Siegenthaler, U., Stocker, T.F., Monnin, E., Lüthi, D., Schwander, J., Stauffer, B., Raynaud, D., Barnola, J., Fischer, H., Masson-Delmotte, V. & Jouzel, J. 2005. Stable carbon cycle–climate relationship during the late pleistocene. *Science*, 310(5752): 1313–1317.

Simberloff, D. 1996. Impacts of introduced species in the United States. *Consequences*, 2(2):13–22.

Singh, B. & Sekhon, G.S. 1976 Nitrate pollution of groundwater from nitrogen fertilizers and animal wastes in the Punjab, India. *Agriculture and Environment*, 3(1): 57–67.

Siriwardena, L., Finlayson, B.L. & McMahon T.A. 2006. The impact of land use change on catchment hydrology in large catchments: The Comet River, Central Queensland, *Australia Journal of Hydrology*, 326(14): 199–214.

Sirohi, S. & Michaelowa, A. 2004. *CDM potential of dairy sector in India.* HWWA discussion paper 273. Hamburgisches Welt-Wirtschafts-Archiv (HWWA). Hamburg Institute of International Economics. 73 pp.

Skinner, B.J., Porter, S.C. & Botkin, D.B. 1999. *The blue planet: An introduction to earth system science*, Second Edition. 576 pp.

Slifko, T.R., Smith, H.V. & Rose, J.B. 2000. Emerging parasite zoonoses associated with water and food. *International Journal for Parasitology*, 30(12–13): 1379–1393.

Small, L. & Carruthers, I. 1991. *Farmer financed irrigation – allocating a scarce resource: Water-use Efficiency* (Chapter 5 pp 77–95), Cambridge University Press.

Smil, V. 1999. Nitrogen in crop production: an account of global flows. *Global Biogeochemical Cycles*, 13(2): 647–662.

Smil, V. 2001. *Enriching the earth: Fritz Haber, Carl Bosch, and the transformation of world food production*. USA, MIT Press. p. 411.

Smil, V. 2002. Nitrogen and food production: proteins for human diets. *Ambio*, 31(2): 126–131.

Smith, B.E. 2002. Nitrogenase reveals its inner secrets. *Science*, 297(5587): 1654–1655.

Sommer, S.G., Petersen, S.O. & Møller, H.B. 2004. Algorithms for calculating methane and nitrous oxide emissions from manure management. *Nutrient Cycling in Agroecosystems*, 69: 143–154.

Soto, A., Calabro, J.M., Prechtl, N.V., Yau, A.Y., Orlando, E.F., Daxenberger, A., Kolok, A.S., Guillette, L.J., Jr., le Bizec, B., Lange, I.G. & Sonnenschein, C. 2004. Androgenic and estrogenic activity in water bodies receiving cattle feedlot effluent in eastern Nebraska, USA . *Environmental Health Perspectives*, 112(3).

Spahni, R., Chappellaz, J., Stocker, T.F., Loulergue, L., Hausammann, G., Kawamura, K., Flückiger, J., Schwander, J., Raynaud, D., Masson-Delmotte, V. & Jouzel, J. 2005. Atmospheric methane and nitrous oxide of the late Pleistocene from Antarctic ice cores. *Science*, 310(5752): 1317–1321.

Speedy, A.W. 2003. Global production and consumption of animal source foods. *Journal of Nutrition*, 133: 4048S–4053S.

Stallknecht, D.E. & Justin D. 2006 *Brown wild birds and the epidemiology of Avian Influenza*. FAO/OIE International Scientific Conference on Avian Influenza and Wild Birds.

Steinfeld, H., de Haan, C.H., & Blackburn, H. 1997. *Livestock and the environment Interactions: Issues and Options*. Suffolk, UK, WRENmedia.

Steinfeld, H. & Chilonda, P. 2006. Old players, new players. *Livestock Report 2006*. Rome, FAO.

Steinfeld, H., Costales, A. & Gerber, P. 2005. Underneath the livestock revolution: Structural change. In: McLeod, A., ed. 2006. *Livestock Report 2006*. Rome, FAO.

Steinfeld, H., Wassenaar, T. & Jutzi, S. 2006. Livestock production systems in developing countries: Status, drivers, trends. *Rev. Sci. Rech. Off. Int. Epiz.*, 25(2). In press.

Stoate, C., Boatman, N.D., Borralho, R.J., Carvalho, C.R., Snoo, G.R.D. & Eden, P. 2001. Ecological impacts of arable intensification in Europe. *Journal of Environmental Management*, 63(4): 337–365.

Sumberg, J. 2003. Toward a dis-aggregated view of crop–livestock integration in Western Africa. *Land Use Policy*, 20(3) 253–264.

Sundermeier, A., Reeder, R. & Lal, R. 2005. *Soil carbon sequestration – Fundamentals*. Ohio State University Extension Fact Sheet AEX-510-05. (Available at ohioline.osu.edu/aex-fact/0510.html).

Sundquist, B. 2003. *Grazing lands degradation: A global perspective*, Edition 4.

Sundquist, E.T. 1993. The global carbon dioxide budget. Science, 259: 934–941.

Sustainable Table. 2005. *The issues, Environment*. (Available at http://www.sustainabletable.org/issues/environment/).

Sutton, A., Applegate, T., Hankins, S., Hill, B., Sholly, D., Allee, G., Greene, W., Kohn, R., Meyer, D., Powers, W. & van Kempen, T. 2001 Manipulation of animal diets to affect manure production, composition and odours: state of the science. pp 377–408. In *animal agriculture and the environment: National Center For Manure And Animal Waste Management White Papers*. J.M. Rice, D.F. Caldwell & F.J. Humenik, eds. St. Joseph, Michigan, USA. Published by the American Society of Agricultural and Biological Engineers.

Swanson, S. 1996. *Riparian pastures*, Fact sheet No. 19. Rangeland Watershed Program; UC Cooperative Extension and USDA Natural Resources Conservation Service; California Rangelands Research and Information Center, Agronomy and Range Science. Davis, USA, University of California.

Swift, M.J., Seward, P.D., Frost, P.G.H., Qureshi, J.N. & Muchena, F.N. 1994. Long-term experiments in Africa: developing a database for sustainable land use under global change. In R.A. Leigh, & A.E. Johnston, eds. *Long-term experiments in agricultural and ecological sciences*, pp. 229–251. Wallingford, UK, CABI Publishers.

Syvitski, J.P.M., Peckham, S.D., Hilberman, R. & Mulder, T. 2003. Predicting the terrestrial flux of sediment to the global ocean: a planetary perspective. *Sedimentary Geology*, 162: 5–24.

Szott, L., Ibrahim, M. & Beer, J. 2000. *The hamburger connection hangover: cattle pasture land degradation and alternative land use in Central America*. Turrialba, Costa Rica, CATIE.

Tabarelli, M. & Gascon, C. 2005. Lessons from fragmentation research: Improving management and policy guidelines for biodiversity conservation. *Conservation Biology*, 19(3): 734–739.

Tadesse, G. & Peden, D. 2003. Livestock grazing impact on vegetation, soil and hydrology in a tropical highland watershed. In: McCornick, P.G., Kamara, A.B., and Tadesse, G., eds. *Integrated water and land management research and capacity building priorities for Ethiopia*. Proceedings of a MoWR/EARO/IWMI/ILRI international workshop held at ILRI, Addis Ababa, Ethiopia 2–4 December 2002, pp. 87–97.

Tansey, Grégoire, J., Stroppiana, D., Sousa, A., Silva, J., Pereira, J.M.C., Boschetti, L., Maggi, M., Brivio, P.A., Fraser, R., Flasse, S., Ershov, D., Binaghi, E., Graetz, D. & Peduzzi, P. 2004. Vegetation burning in the year 2000: Global burned area estimates from SPOT VEGETATION data. *Journal of Geophysical Research Atmospheres*, VOL. 109, D14S03.

Tate, K.W. 1995. *Infiltration and overland flow*. Fact Sheet No. 37. Rangeland Watershed Program, U.C. Cooperative Extension and USDA Natural Resources Conservation Service; California Rangelands Research and Information Center, Agronomy and Range Science, Davis, USA, University of California.

Thellung A. 1912. La flore adventice de Montpellier. *Memoires de la Société Nationale des Sciences Naturelles et Mathématiques*, 38: 57–728.

The State of Queensland. 2004. *The Desert Uplands*. NRM facts, Land series. The State of Queensland, Department of Natural Resources and Mines.

Thobani, M. 1997. Formal water markets: why, when and how to introduce tradable water rights. *The World Bank Research Observer*, 12(2), August 1997: 163–165.

Thomas, C.D., Cameron, A., Green, R.E., Bakkenes, M., Beaumont, L.J., Collingham, Y.C., Erasmus, B.F.N., Ferreira de Siqueira, M., Grainger, A., Lee Hannah, Hughes, L., Huntley, B., van Jaarsveld, A.S., Midgley, G.F., Miles, L., Ortega-Huerta, M.A., Peterson, A.T., Phillips, O.L. & Williams, S.E. 2004. Extinction risk from climate change. *Nature*, 427:145–148.

Thomas, D.S. 2002. Sand, grass, thorns, and cattle: The modern Kalahari. In: Deborah Sporton and David S. G. Thomas, eds., *Sustainable livelihoods in Kalahari environments: Contributions to global debates.* New York, Oxford University Press, Inc. pp 21–38.

Thompson, A.M., Witte, J.C., Hudson, R.D., Guo, H, Herman, J.R. & Fujiwara, M. 2001. Tropical tropospheric ozone and biomass burning. *Science*, 291: 2128–2132.

Thorsten, C., Schneider, R.J., Färber, H.A., Skutlarek, D. & Goldbach, H.E. 2003. Determination of antibiotic residues in manure, soil, and surface waters. *Acta hydrochimica et hydrobiologica,* 31(1): 36–44.

Tidwell, J.H. & Allan, G.L. 2001. *Fish as food: aquaculture's contribution*. European Molecular Biology Organization (EMBO) reports 2(11): 958–963.

Tietenberg, T. 2003. *Environmental and natural resource economics*, sixth edition, Addison Wesley.

Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D., Schlesinger, W.H., Simberloff, D. & Swackhamer, D. 2001. Forecasting agriculturally driven global environmental change. *Science*, 292(5515): 281–284.

Tonhasca, A. & Byrne, D.N. 1994. The effects of crop diversification on herbivorous insects: a meta-analysis approach. *Ecological Entomology*, 19: 239–244.

Toutain, B. 2001. *Mission d'appui scientifique sur le thème de la transhumance*. Programme régional parc du W - ECOPAS. Rapport CIRAD – EMVT No.01-43, July 2001.

Tran Thi Dan, Thai Anh Hoa, Le Quang Hung, Bui Minh Tri, Ho Thi Kim Hoa, Le Thanh Hien & Nguyen Ngoc Tri. 2003. *LEAD pilot project on the Area-wide integration (AWI) of specialized crop and livestock activities in Vietnam.* C. Narrod & P. Gerber, eds.

Travasso, M.I., Magrin, G.O., Rodríguez, G.R. & Boullón, D.R. 1999. Climate Change assessment in Argentina: II. Adaptation strategies for agriculture. Accepted in *Food and Forestry: Global Change and Global Challenge*. GCTE Focus 3 Conference. Reading, United Kingdom, September 1999.

Tremblay, L.A. & Wratten, S.D. 2002. Effects of Ivermectin in dairy discharges on terrestrial and aquatic invertebrates. *DOC Science Internal Series 67*. Department of Conservation, Wellington. 13 pp.

Trimble, S.W. & Mendel, A.C. 1995. The cow as a geomorphic agent—a critical review. *Geomorphology*, 13: 233–253.

Tschakert, P. & Tappan, G. 2004. The social context of carbon sequestration: considerations from a multi-scale environmental history of the Old Peanut Basin of Senegal. *Journal of Arid Environments*, 59(3): 535–564.

Tsur, Y. & Dinar, A. 1997. The relative efficiency and implementation costs of alternative methods of pricing irrigation water. *The World Bank Economic Review*, 11(2), May 1997: 243–262.

Turner, K., Georgiou, S., Clark, R., Brouwer, R. & Burke, J. 2004. Economic Valuation of water resources in agriculture, From the sectoral to a functional perspective of natural resource management. FAO paper reports No. 27, Rome, FAO.

Tveteras, S. & Tveteras, R. 2004. *The global competition for wild fish resources between livestock and aquaculture*. 13th Annual Conference of the European Association of Environmental and Resource Economics. 25–28 June 2004, Budapest, Hungary.

UN. 1992. *Rio Declaration on Environment and Development*, New York, USA, United Nations Conference on Environment and Development (UNCED) 1992.

UN. 2003. *World population prospects: The 2002 revision*, United Nations, New York, USA.

UN. 2005. *World Population Prospects*. The 2004 Revision. UN Department of Economic and Social Affairs. New York, USA, (Available at http://www.un.org/esa/population/publications/sixbillion/sixbilpart1.pdf).

UNDP/UNEP/WB/WRI. 2000. *A guide to world resources 2000–2001: People and ecosystems – The fraying web of life.* United Nations Development Programme, United Nations Environment Programme, World Bank, World Resources Institute. Washington, DC, World Resources Institute.

UNEP. 1991. *Status of desertification and implementation of the United Nations Plan of Action to combat desertification*. United Nations Environment Programme, Nairobi. 79 pp.

UNEP. 1994. Land degradation in South Asia: Its severity, causes and effects upon the people. World Soil Resources Report 78. INDP/UNEP/FAO. Rome, FAO.

UNEP. 1997. *World atlas of desertification*. United Nations Environment Programme, Second Edition. Nairobi. 182 pp.

UNEP. 2001. *Economic reforms, trade liberalization and the environment: A Synthesis of UNEP Country Projects.* Geneva, Switzerland. 5 November 2001.

UNEP. 2002. *Protecting the environment from land degradation* UNEP's action in the framework of the Global Environment Facility.

UNEP. 2003. *Global Environmental Outlook 3*. UNEP GEO Team, Division of Environmental Information, Assessment and Early Warning (DEIA&EW), United Nations Environment Programme.

UNEP. 2004a. *GEO Yearbook 2003*. Earthprint, 76 p. (Available at www.unep.org/geo/yearbook/yb2003/).

UNEP. (2004b). *Land degradation in drylands (LADA): GEF grant request.* Nairobi, Kenya: UNEP.

UNEP. 2005a. *Global Environment Outlook Year Book 2004/05,* (Available at http://www.unep.org/geo/yearbook/yb2004/).

UNEP. 2005b. Environmental strategies and policies for cleaner production, (Available at www.uneptie.org/pc/cp/understanding_cp/cp_policies.htm; Accessed November 2005).

UNEP-WCMC. 1994. *Biodiversity data sourcebook.* B. Groombridge, ed. WCMC Biodiversity Series No. 1. World Conservation Monitoring Centre, United Nations Environment Programme, Cambridge, UK, World Conservation Press. 155 pp.

UNEP-WCMC. 1996. *The diversity of the seas: a regional approach*. WCMC Biodiversity Series No. 4. Edited by B. Groombridge and M.D. Jenkins. World Conservation Monitoring Centre, United Nations Environment Programme. Cambridge, UK, World Conservation Press.

UNEP-WCMC. 2000. *Chapter 1 of the Global Biodiversity Outlook: Global Biodiversity: Earth's living resources in the twentifirst century.* B. Groombridge & M.D. Jenkins, Cambridge, UK, World Conservation Press.

UNEP-WCMC. 2002. *Mountain watch: environmental change and sustainable development in mountains*. UNEP-WCMC Biodiversity Series No. 12 World Conservation Monitoring Centre, United Nations Environment Programme, Cambridge, UK, World Conservation Press.

UNESCO. 1979. *Tropical grassland ecosystems*. UNESCO, Paris.

UNESCO. 2005. Water portal. (Available at http://www.unesco.org/water/).

UNFCCC. 2005. *Feeling the heat*. Electronic background document - United Nations Framework Convention on Climate Change, (Available at http://unfccc.int/essential_background/feeling_the_heat/items/2918.php).

Uri, N.D. & Lewis, J.A. 1998. The dynamics of soil erosion in US agriculture. *The Science of the Total Environment*, 218(1): 45–58.

USDA. 2004. *US agriculture and forestry greenhouse gas inventory: 1990–2001*. U.S. Department of Agriculture, Global Climate Change Program, Technical Bulletin No. 1907. (Available at www.usda.gov/oce/global_change/gg_inventory.htm).

USDA/ERA. 2002. *Adoption and pesticide use. In: Adoption of Bioengineered Crops*. Agricultural Economic Report No. (AER810) 67 pp.

USDA/FAS. 2004. *World Broiler Trade Overview*. Foreign Agricultural Service, United States Department of Agriculture.

USDA/FAS. 2000. *Meat and bone meal ban may induce South American soybean planting*. United States Department of Agriculture: Foreign Agricultural Service, (Available at www.fas.usda.gov/pecad2/highlights/2000/12/EU_mbm_ban.htm).

USDA/NASS. 2001. *Agricultural chemical use*. National Agricultural Statistics Service, USDA Economics and Statistics System.

USDA/NRCS. 1998. *Soil quality indicators: Infiltration*. Soil Quality Information Sheet. Soil Quality Institute, Natural Resources Conservation Service (NRCS), US Department of Agriculture (USDA).

USDA-NRCS. 1999. *Risk of human induced water erosion map.*, Soil Survey Division, World Soil Resources. Washington, DC, USDA-NRCS.

US-EPA. 2004. *US emissions inventory 2004: Inventory of u.s. greenhouse gas emissions and sinks: 1990–2002* (April 2004). US Environmental Protection Agency.

US-EPA. 2005a. Mid-Atlantic Integrated Assessment (MAIA), (Available at http://www.epa.gov/maia/html/about.html).

US-EPA. 2005b. *Global warming – Methane*. US Environmental Protection Agency, (Available at http://www.epa.gov/methane/).

US-EPA. 2006. *EPA livestock analysis model*, (Available at http://www.epa.gov/methane/rlep/library/lam/lam.html).

US-Geological Survey. 2005a. *The water cycle*. United States Geological Survey (USGS). (Available at http://ga.water.usgs.gov/edu/watercycle.html).

US-Geological Survey. 2005b. *Estimated use of water in the United States in 1990: Livestock water use*. United States Geological Survey (USGS), (Available at http://water.er.usgs.gov/watuse/wulv.html).

Van Aardenne, J.A., Dentener, F.J., Olivier, J.G.J., Klein Goldewijk, C.G.M. & J. Lelieveld. 2001. A high resolution dataset of historical anthropogenic trace gas emissions for the period 1890–1990. *Global Biogeochemical Cycles*, 15(4): 909–928.

van Auken, W.O. 2000. Shrub invasions of North American semi-arid grasslands. *Annual Review Of Ecology and Systematics*, 31:197–216.

Van der Hoek, K.W. 1998. Nitrogen efficiency in global animal production. *Environmental Pollution*, 102: 127–132.

van Ginkel, J.H., Whitmore, A.P. & Gorissen, A. 1999. Lolium perene grasslands may function as a sink for atmospheric carbon dioxide. *Journal of Environmental Quality*, 28: 1580–1584.

Van Vuuren, A.M. & Meijs, J.A.C. 1987, In: *Animal manure on grassland and fodder crops: fertiliser or waste?* H.G.Van der Meer, R.J. Unwin, T.A. van Dijk & G.C. Ennik, eds., Nijhoff, Dortrecht, the Netherlands. pp 27–45.

Vannuccini, S. 2004. Overview of fish production, utilization, consumption and trade. Fishery Information, Data And Statistics Unit, FAO. 20 pp.

Vasilikiotis, C. 2001. *Can Organic Farming "Feed the World"?* University of California, Berkeley. Energy Bulletin, published 1 February 2001. (Available at http://www.energybulletin.net/1469.html).

Velusamy, R., Singh, B.P. & Raina, O.K. 2004. Detection of *Fasciola gigantica* infection in snails by polymerase chain reaction. *Veterinary Parasitology*, 120(1-2): 85–90.

Vera, F.W.M. 2000. *Grazing ecology and forest history*. Wallingford, UK, CABI Publishing.

Verburg, P.H., Chen, Y.Q. & Veldkamp, A. 2000. Spatial explorations of land-use change and grain production in China. Agriculture, Ecosystems and Environment, 82: 333–354.

Verburg, P.H. and Hugo, A.C. & Van Der Gon, D. 2001. Spatial and temporal dynamics of methane emissions from agricultural sources in China. *Global Change Biology*, 7(1): 31–47.

Vet, R. 1995. *GCOS observation programme for atmospheric constituents: Background, status and action plan*. Global Climate Observing System Report No. 20. World Meteorological Organization, (Available at http://www.wmo.ch/web/gcos/Publications/gcos-20.pdf).

Vickery, J. A., Tallowin, J.R.B., Feber, R.E., Asteraki, E.J., Atkinson, P.W., Fuller, R.J. & Brown, V.K. 2001. The management of lowland neutral grasslands in Britain: effects of agricultural practices on birds and their food resources. *Journal of Applied Ecology*, 38: 647–664.

Viollat, P.L. 2006. *Argentine, un cas d'école*. Le Monde Diplomatique, April 2006.

Vitousek, P.M., Aber, J.D., Howarth, R.W., Likens, G.E., Matson, P.A., Schindler, D.W., Schlesinger, W.H. & Tilman, D.G. 1997 Human alteration of the global nitrogen cycle: sources and consequences. *Ecological Applications*, 7(3): 737–750.

Vlek, P.L.G., Rodríguez-Kuhl G. & Sommer R. 2004. Energy use and CO_2 production in tropical agriculture and means and strategies for reduction or mitigation. *Environment, Development and Sustainability*, 6(1-2): 213–233.

von Dörte, E. 2004. *Water management in rural China: The role of irrigation water charges.*

von Tschudi, J.J. 1868. *Reisen durch Sudamerika*, Vol. 4. Leipzig: Verlag 1 Brockhaus. Stuttgrt: Omnitypie-Gesellshaft Nachf. Leopold Zechnall rep. 1971. 320 pp.

Vought, L.B.M., Pinay, G., Fuglsang, A. & Ruffinoni, C. 1995. Structure and function of buffer strips from a water quality perspective in agricultural landscapes. *Landscape and Urban Planning*, 31(1–3): 323–331.

Wahl, R.W. 1997. *Water pricing experiences: An international perspective – United States*. World Bank Technical Paper No. 386, pp 144–148.

Walker, R. 1993. Deforestation and economic development. *Canadian Journal of Regional Science*, 16(3): 481–497.

Wallinga, D. 2002. Antimicrobial use in animal feed: An ecological and public health problem. *Minnesota Medical Association*, Volume 85, October 2002.

Ward, A.D. 2004. ACSM 370 Teaching Materials: Principles of Hydrology, Lecture 2: *Infiltration*. Department of Food, Agricultural and Biological Engineering; USA, Ohio State University.

Ward, F. & Michelsen, A. 2002. The economic value of water in agriculture: concepts and policy applications. *Water Policy*, 4(5): 423–446.

Ward, G.M., Knox, P. L. & Hobson, B.W. 1977. Beef production options and requirements for fossil fuel. *Science*, 198: 265–271.

Ward, R.C. & Robinson, M. 2000. *Principles of hydrology* (4th ed.). McGraw-Hill Publishing Company, London, 450 pp.

Wardrop Engineering. 1998. Cited by UNEP Working Group for Cleaner Production in the Food Industry, 2004. Fact Sheet 7: Food Manufacturing Series. (Available at http://www.gpa.uq.edu.au/CleanProd/Res/facts/FACT7.HTM).

Wassenaar, T., Gerber, P., Verburg, P.H., Rosales, M., Ibrahim, M. & Steinfeld, H. 2006. Projecting land use changes in the Neotropics The geography of pasture expansion into forest. *Global Environmental Change*, In press.

Waters, T.F. 1995. *Sediment in streams: sources, biological effects, and control*. American Fisheries Society Monograph 7. American Fisheries Society, Bethesda, Maryland. pp. 251.

Watson, R. & Pauly, D. 2001. Systematic distortions in world fisheries catch trends. *Nature*, 414: 534 536.

Webster, R.G., Naeve, C.W. & Krauss, S. 2006. *The evolution of influenza viruses in wild birds*. FAO/OIE International Scientific Conference on Avian Influenza and Wild Birds.

Westing, A.H., Fox, W. & Renner, M. 2001. *Environmental degradation as both consequence and cause of armed conflict*. Working Paper prepared for Nobel Peace Laureate Forum participants by PREPCOM subcommittee on Environmental Degradation.

White, R.P., Murray, S. & Rohweder, M. 2000. *Pilot analysis of global ecosystems: grassland ecosystems*. Washington, DC, World Resources Institute.

Whitmore, A.P. 2000. *Impact of livestock on soil*. Sustainable Animal Production, (Available at http://www.agriculture.de/acms1/conf6/ws4lives.htm).

WHO. and Tufts University. 1998. Keeping fit for life: Meeting the nutritional needs of older persons.

WHO. 2003. Obesity and overweight. Fact sheet, Geneva, World Health Organization

Wichelns, D. 2003. The role of public policies in motivating virtual water trade, with an example from Egypt (I. Delft, Trans.). In A.Y. Hoekstra, ed., 2003. *Virtual water trade: Proceedings of the international expert meeting on virtual water trade*.

Wilcock, R.J., Scarsbrook. M.R., Cooke, J.G., Costley, K.J. & Nagels, J.W. 2004. Shade and flow effects on ammonia retention in macrophyte-rich streams: implications for water quality. *Environmental Pollution*, 132(1): 95–100.

Woodroffe, R., Linsey, P., Romanach, S., Stein, A. & Ranah, S. M.K. 2005. Livestock predation by endangered African wild dogs (*Lycaon pictus*) in northern Kenya. *Biological Conservation*, 124: 225–234.

World Bank. 1997. *Water pricing experiences An international perspective*. World Bank Technical Paper No. 386, Washington, DC, World Bank.

World Bank. 2005a. Managing the livestock revolution: Policy and technology to address the negative impacts of a fast-growing sector. Washington, DC, World Bank Agriculture and Rural Development Department.

World Bank. 2005b. *World development indicators: poverty estimates*.

World Bank. 2006. *World development indicators*. Washington, DC.

World Conservation Monitoring Centre. 1998. *Freshwater biodiversity: a preliminary global assessment*. By Brian Groombridge and Martin Jenkins. WCMC Cambridge, UK, World Conservation Press.

World Resources Institute. 2000. Freshwater biodiversity in crisis.

World Resources Institute. 2003. *The watersheds of the world CD*. Published by IUCN – The World Conservation Union, the International Water Management Institute (IWMI), the Ramsar Convention Bureau, and the World Resources Institute (WRI).

World Resources Institute. 2005. *EarthTrends: The Environmental Information Portal.* (Available at http://earthtrends.wri.org. Accessed 10/12/2005).

WWF. 2003. Soy expansion losing forests to the fields. Forest Conversion INFO – Soy. WWF Forest Conversion Initiative (Available at http://assets.panda.org/downloads/wwfsoyexpansion.pdf#search=%22WWF%202003%20-%20CERRADO%22).

WWF. 2005. *Wild Places Ecoregions*. (Available at http://www.worldwildlife.org/ecoregions/. Accessed August 2005).

Xercavins Valls, J. 1999. *Carrying capacity in east sub-Saharan Africa: A multilevel integrated assessment and a sustainable development approach*. Doctoral thesis. Universita Politècnica de Catalunya.

Yang, H., Zhang, X. & Zehnder, A.J.B. 2003. Water scarcity, pricing mechanism and institutional reform in Northern China irrigated agriculture. *Agricultural Water Management*, 61: 143–161.

Yang, X., Zhang, K., Jia, B. & Ci, L. 2005. Desertification assessment in China: an overview. *Journal of Arid Environments*, 63(2): 517–531.

You, L., Wood, S. & Wood-Sichra, U. 2006. Generating global crop distribution maps: from census to grid. Contributed paper prepared for presentation at the International Association of Agricultural Economists Conference, Gold Coast, Australia, 11–18 August 2006.

Zhang, H., Dao, Thanh H., Wallace, H.A., Basta, N.T., Dayton, E.A. & Daniel T.C. 2001. *Remediation techniques for manure nutrient loaded soils*. White paper summaries, National center for manure and waste management.

Zhang, H.C., Cao, Z.H., Shen, Q.R. & Wong, M.H. 2003. Effect of phosphate fertilizer application on phosphorus (P) losses from paddy soils in Taihu Lake Region I. Effect of phosphate fertilizer rate on P losses from paddy soil. *Chemosphere*, 50(6): 695–701.

Zhang, Y.K. & Schilling, K.E. 2005. Increasing streamflow and baseflow in Mississippi River since the 1940s: Effect of land use change. *Journal of Hydrology*, In Press, Corrected Proof, Available online 2 December 2005.

Zhou, Z.Y., Wu, Y.R. & Tian, W.M. 2003. Food consumption in rural China: preliminary results from household survey data. Proceedings of the 15th Annual Conference of the Association from Chinese Economics Studies, Australia.

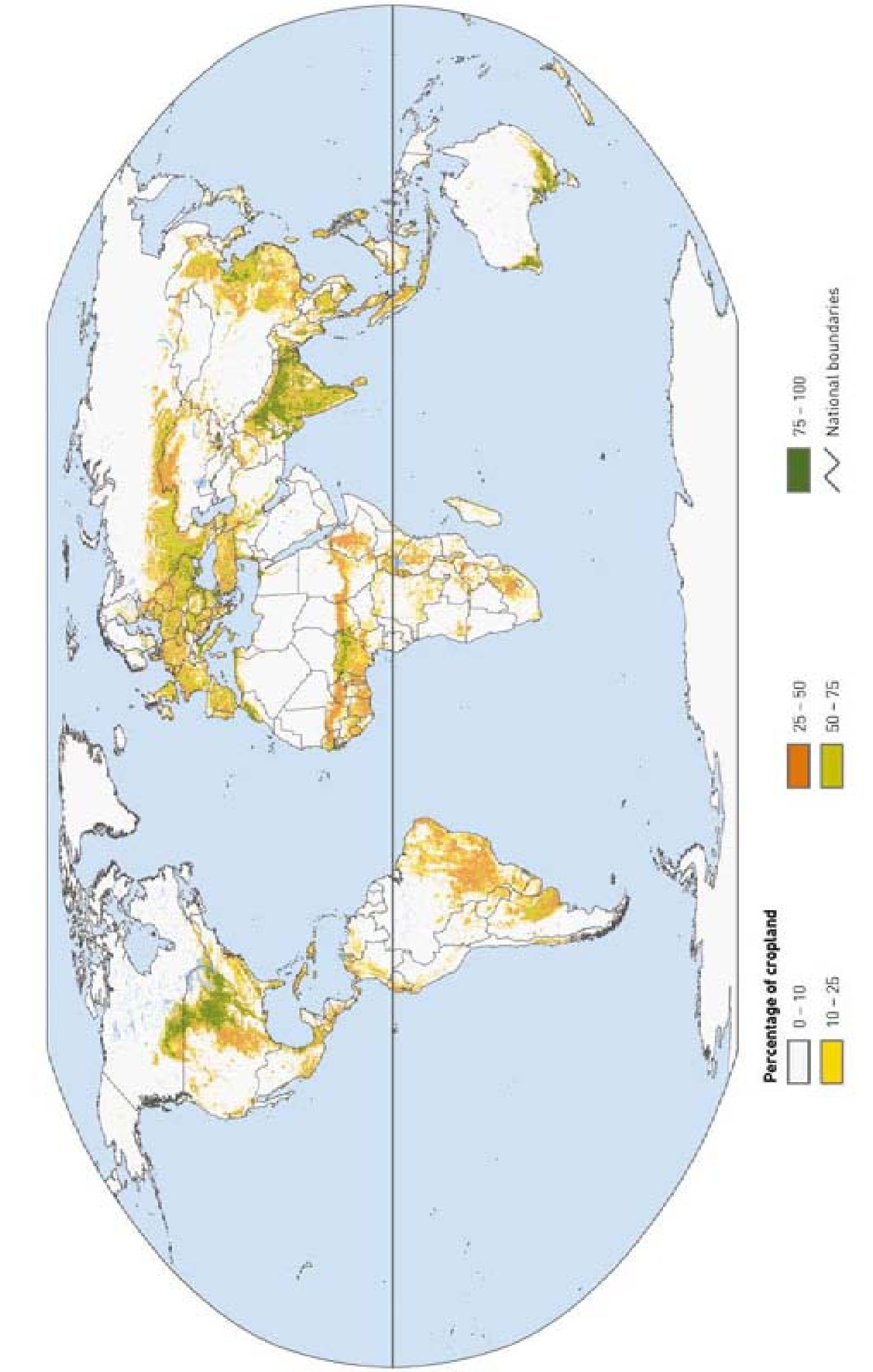
Zimmer, D. & Renault, D. 2003. *Virtual water in food production and global trade: review of methodological issues and preliminary results*. Proceedings of the expert meeting held 12–13 December 2002, Delft, the Netherlands. Editor Arjen Hoekstra, Delft, the Netherlands, 2003, UNESCO-IHE.

Annex 1

Global maps

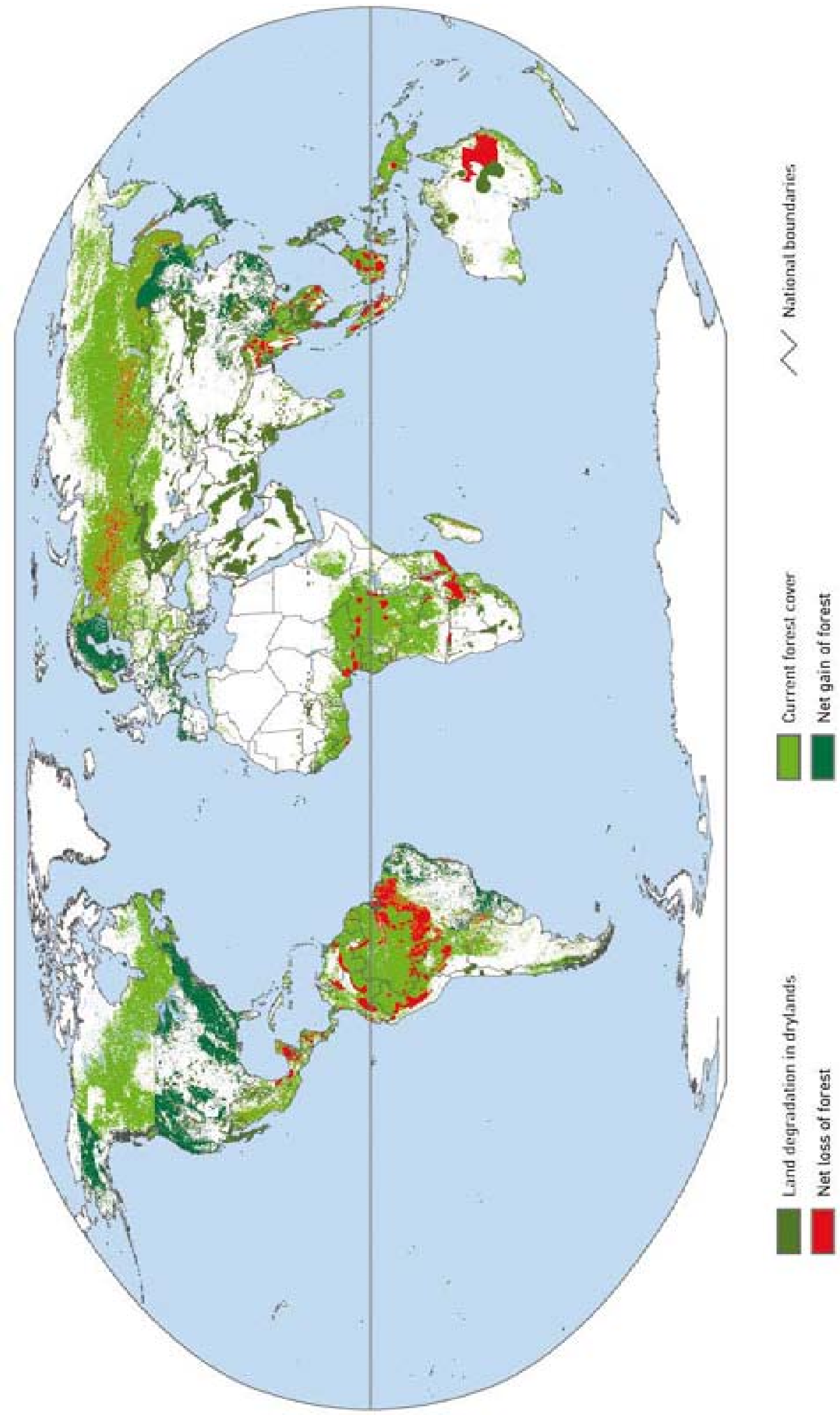
Annex 1
Global maps

Map 1 Extent of cropland in 2000



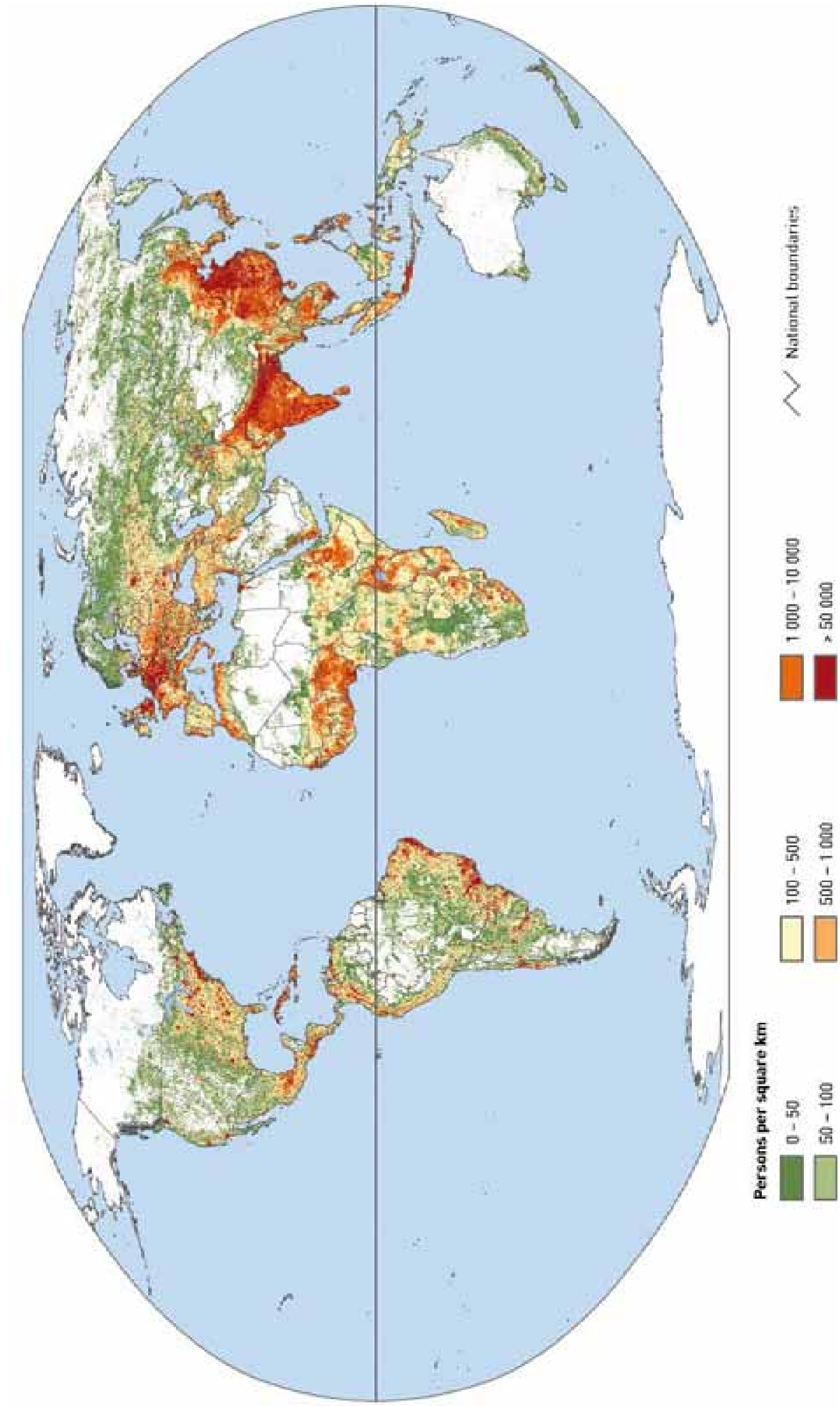
Source: FAO, 2006f.

Map 2 **Forest transition and land degradation in dry lands**



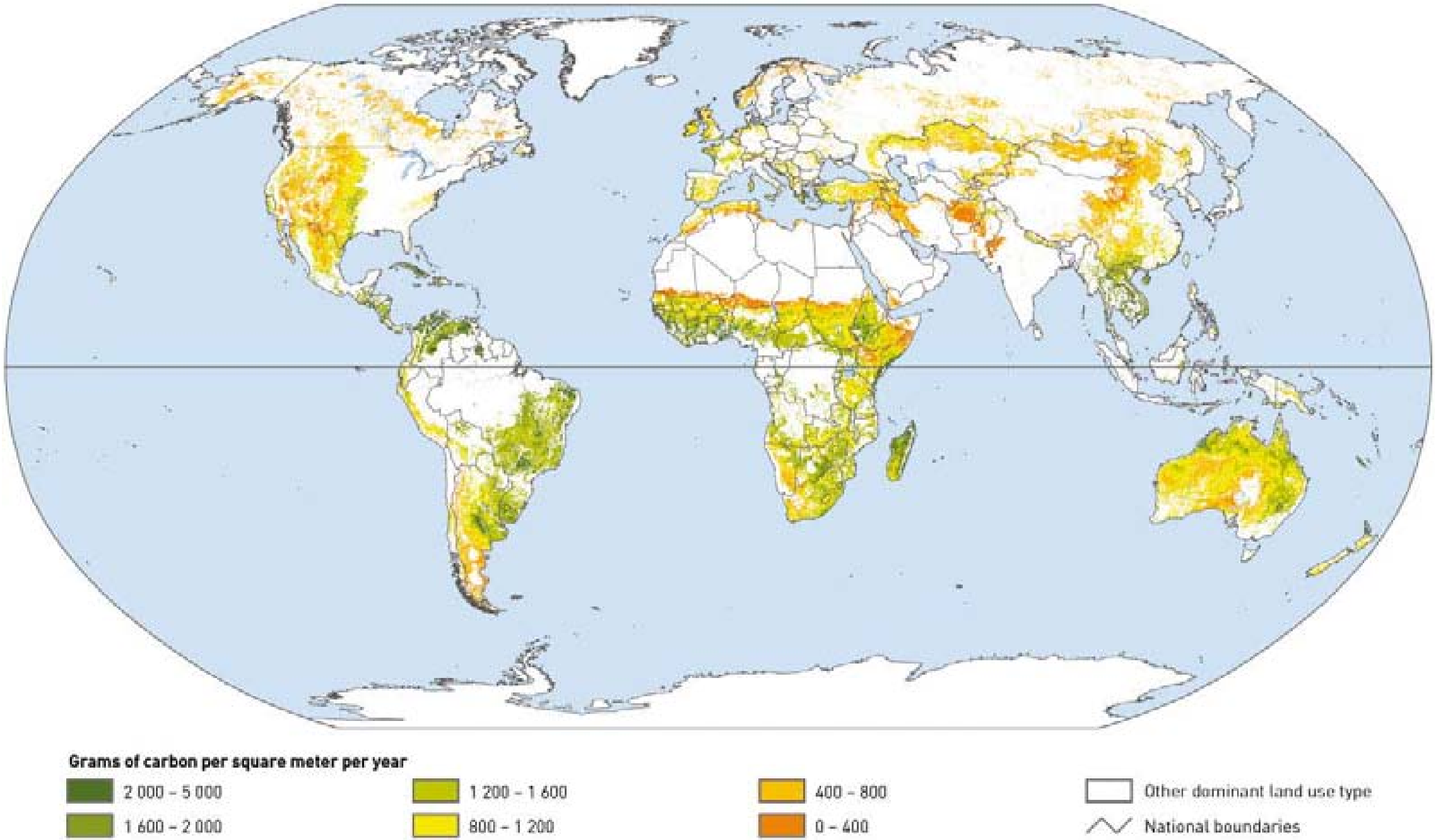
Source: MEA, 2005a.

Map 3 Estimated distribution of human population



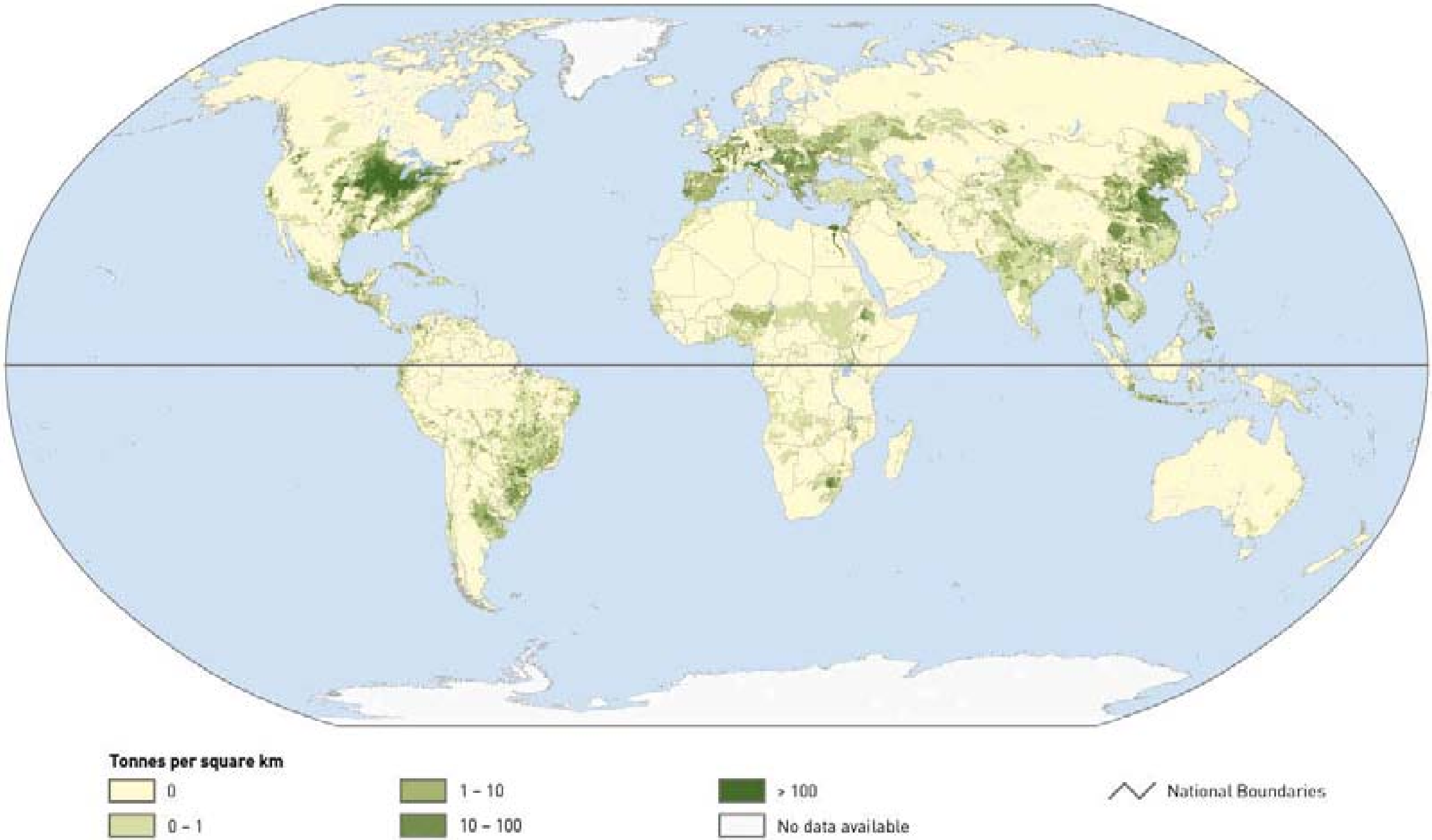
Source: LandScan, 2005.

Map 4 **Estimated net primary productivity in areas dominated by pasture**



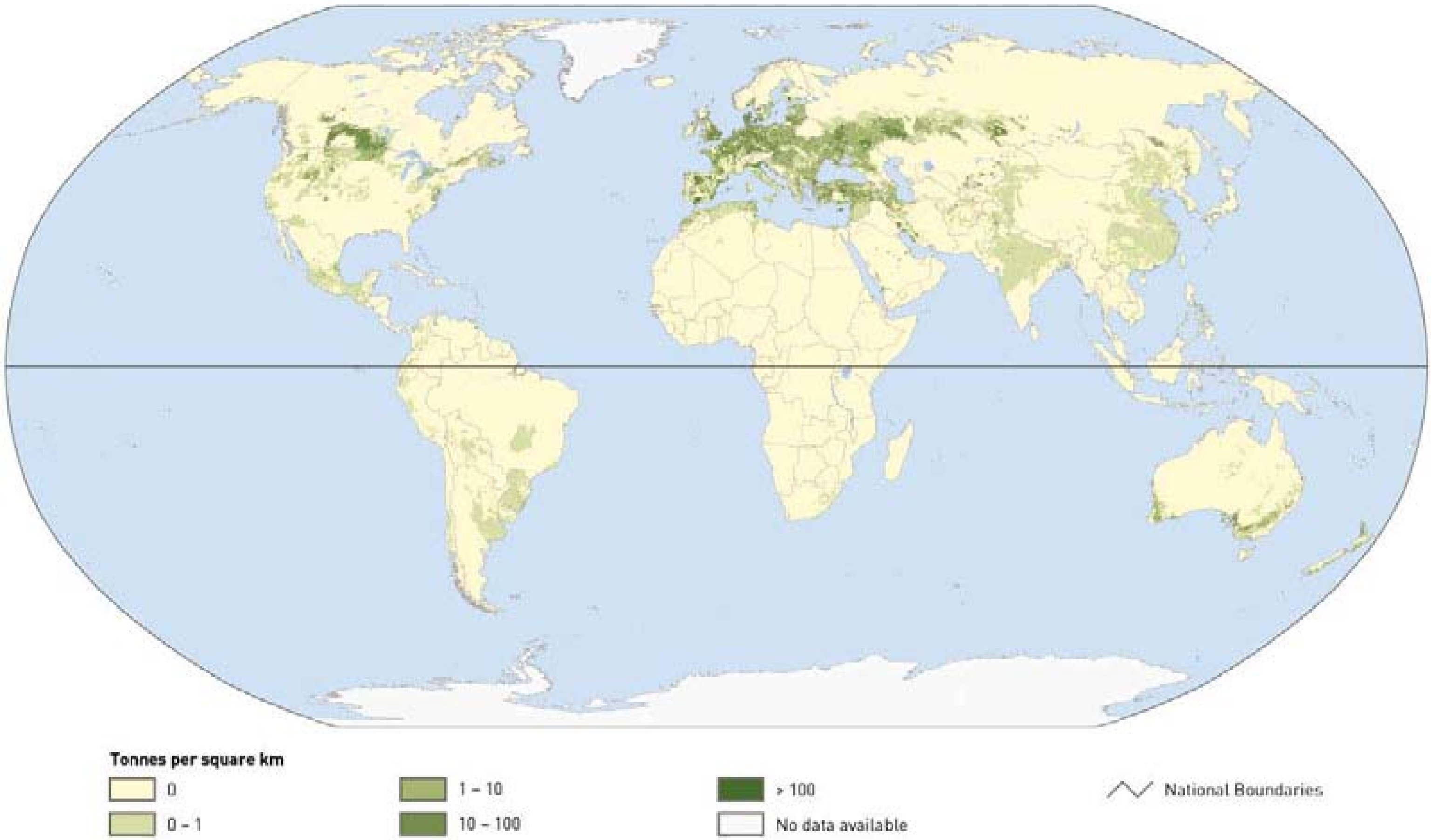
Source: Estimated net primary productivity (Prince and Goward, 1995) is displayed in cells for which at least one-third of the area is used as pasture (FAO,2006f).

Map 5 Estimated maize production for animal feed



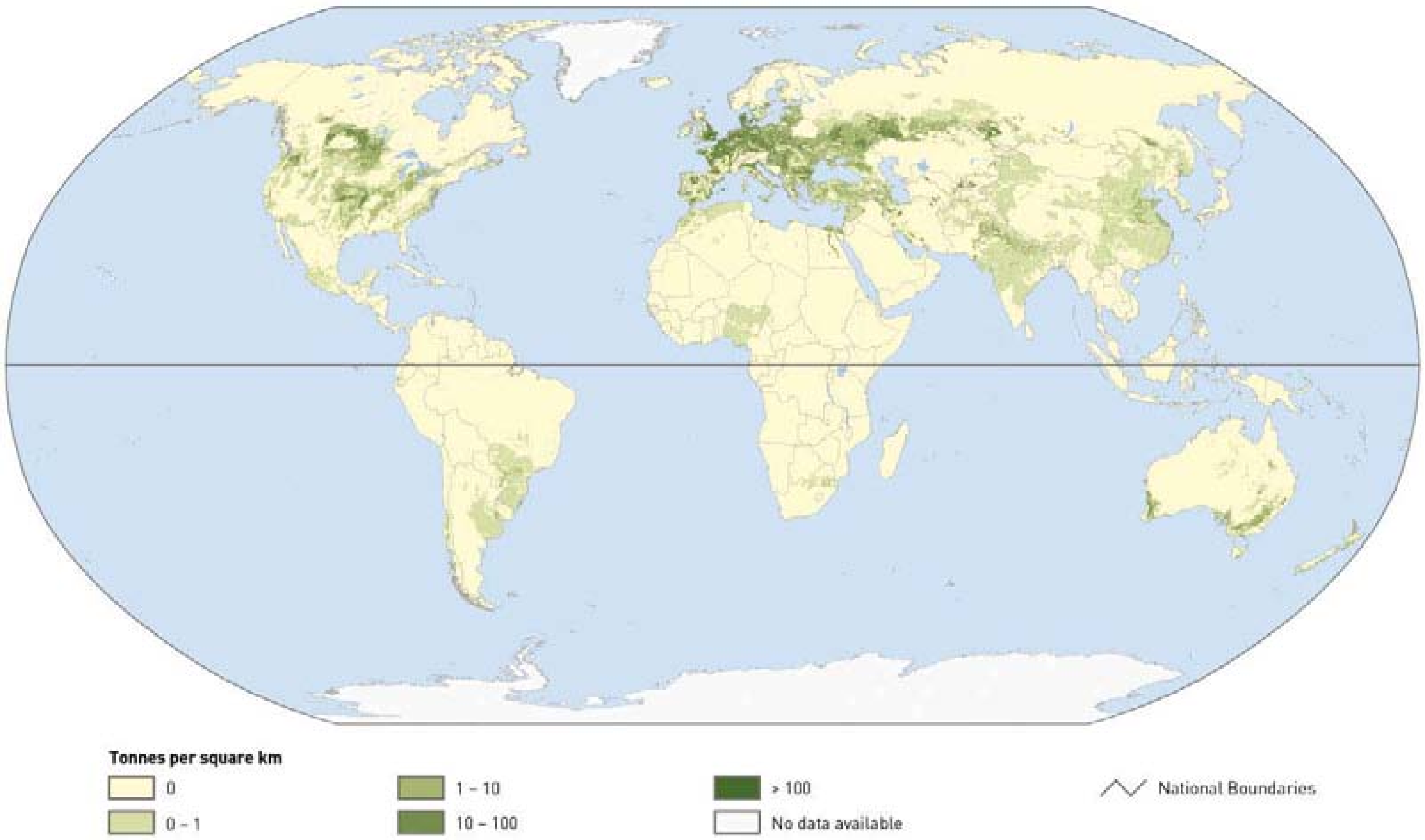
Source: LEAD. The fraction of total production dedicated to feed was estimated at national level (FAO, 2006b) and the ratios applied to the crop production map (You *et al.*, 2006).

Map 6 **Estimated barley production for animal feed**



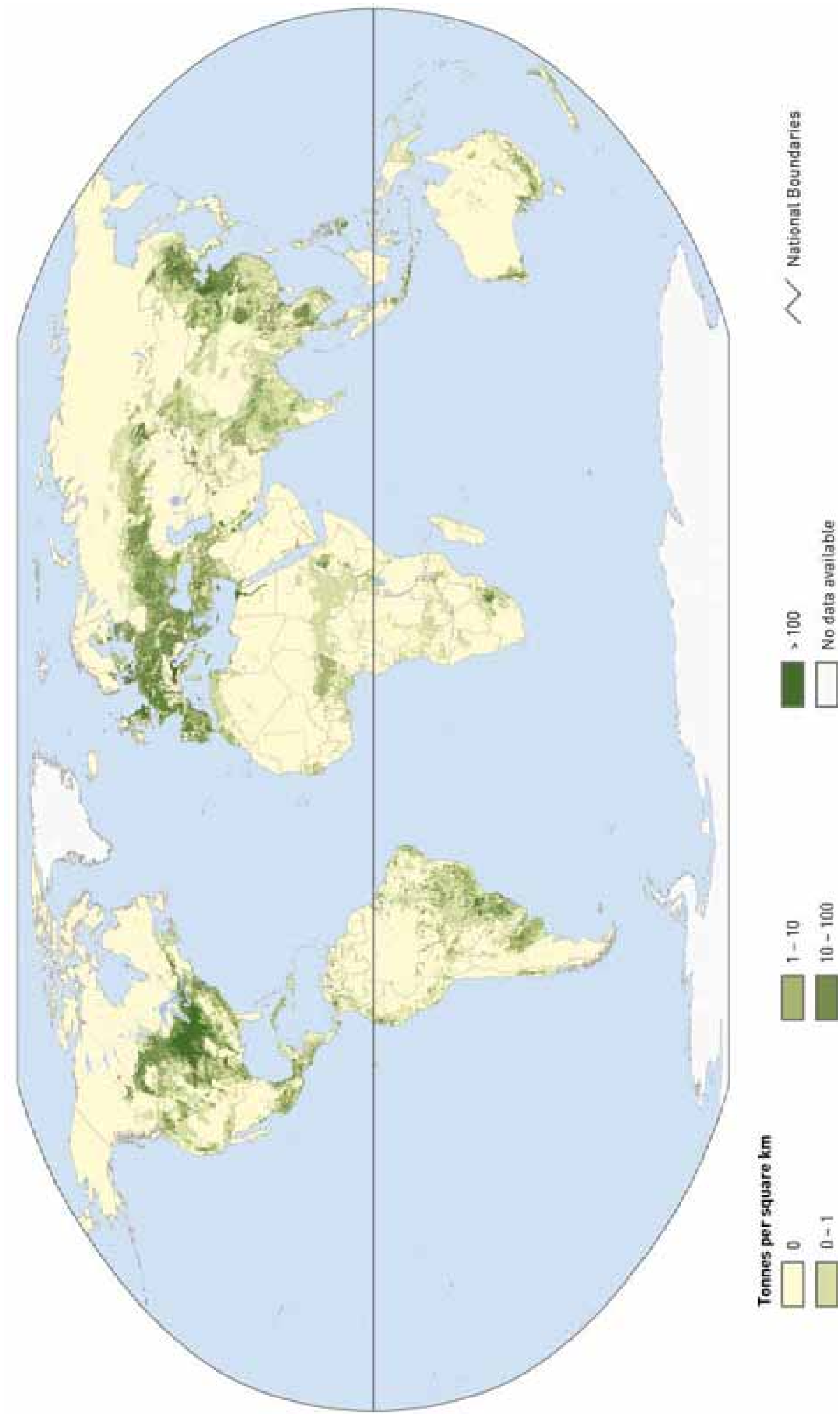
Source: LEAD. The fraction of total production dedicated to feed was estimated at national level (FAO, 2006b) and the ratios applied to the crop production map (You *et al.*, 2006).

Map 7 Estimated wheat production for animal feed



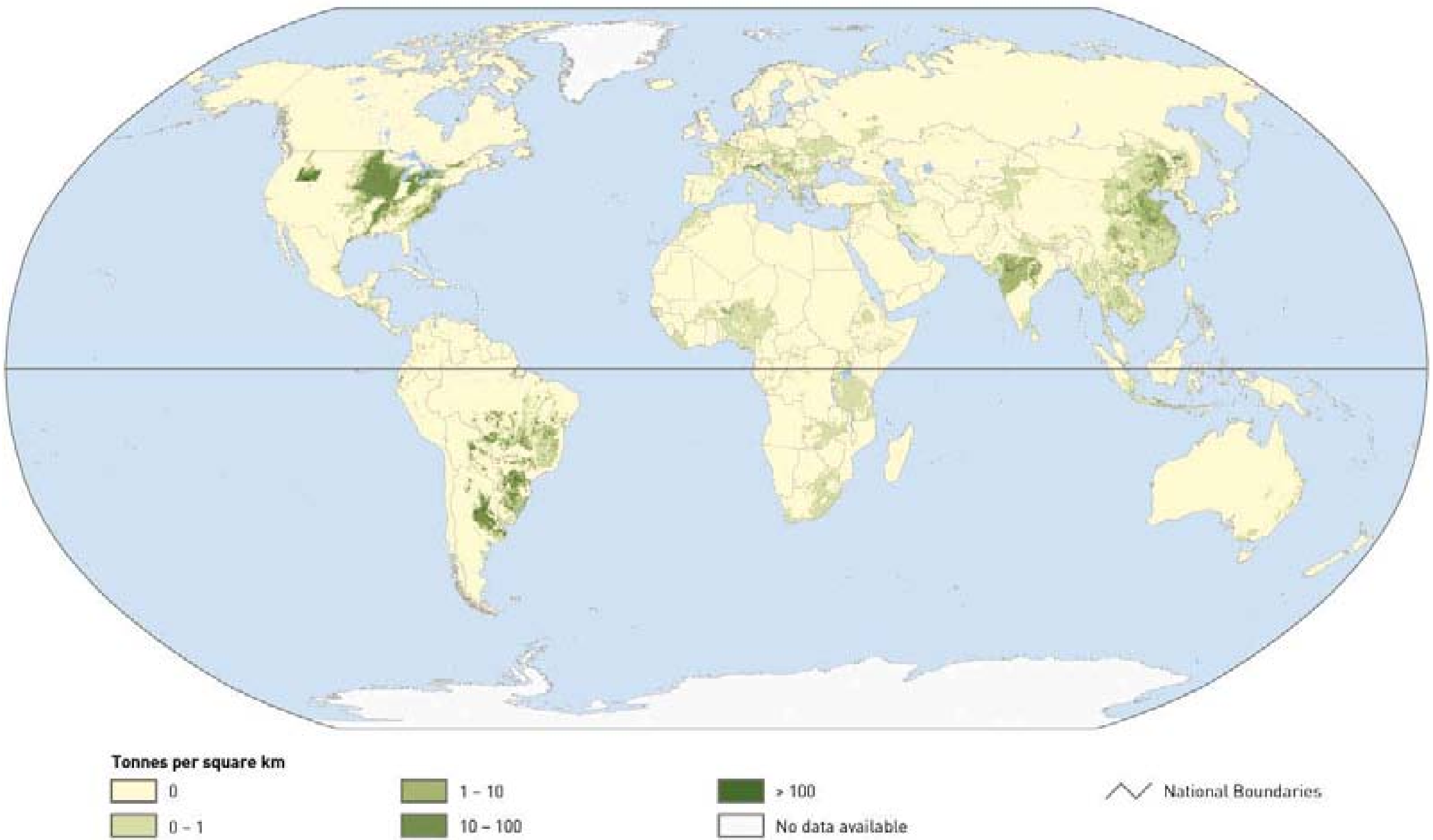
Source: LEAD. The fraction of total production dedicated to feed was estimated at national level (FAO, 2006b) and the ratios applied to the crop production map (You *et al.*, 2006).

Map 8 **Estimated cumulated maize, wheat and barley production for animal feed**



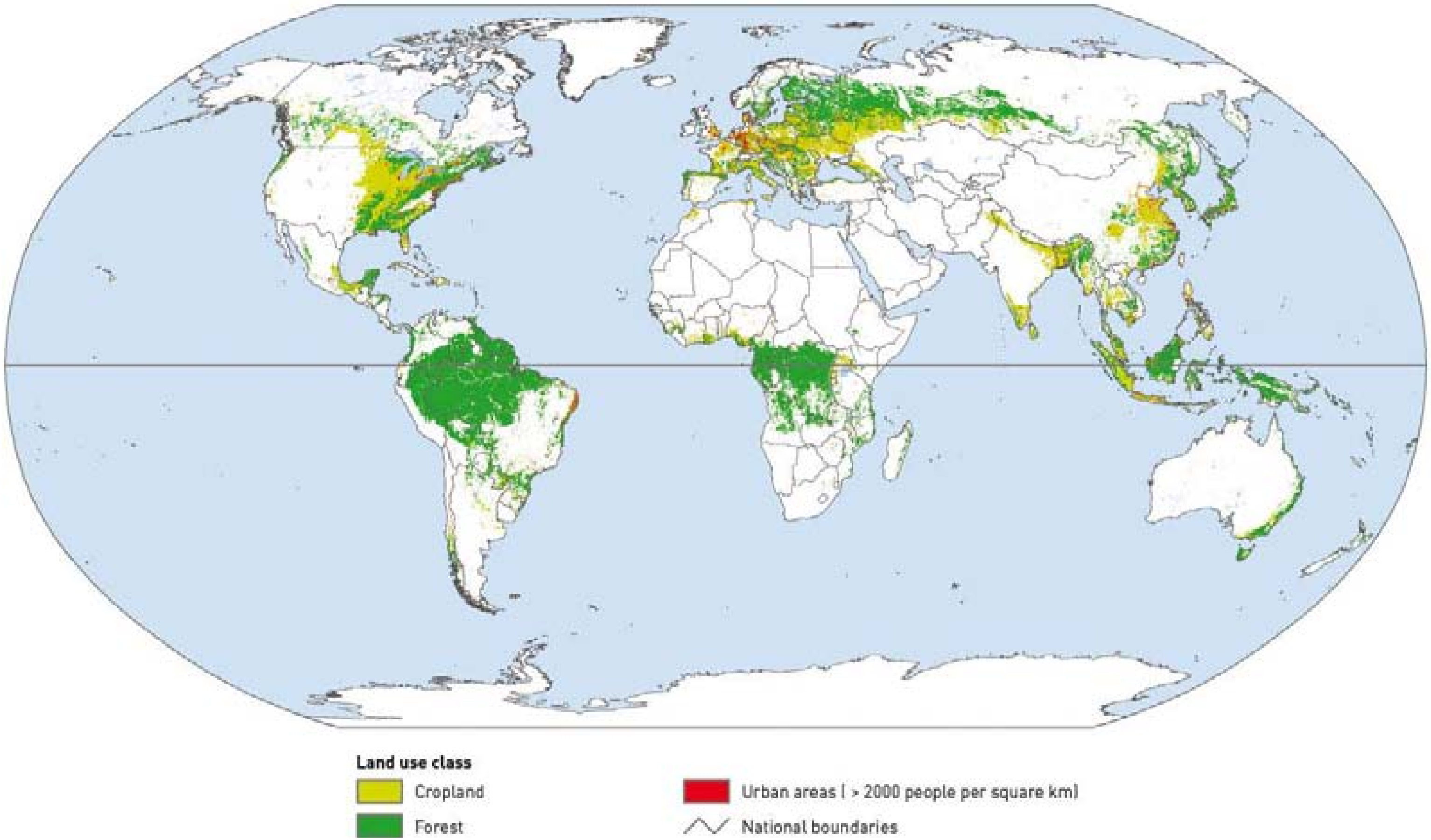
Source: LEAD. The fraction of total production dedicated to feed was estimated at national level (FAO, 2006b) and the ratios applied to the crop production map (You *et al*., 2006).

Map 9 **Estimated soybean production for animal feed**



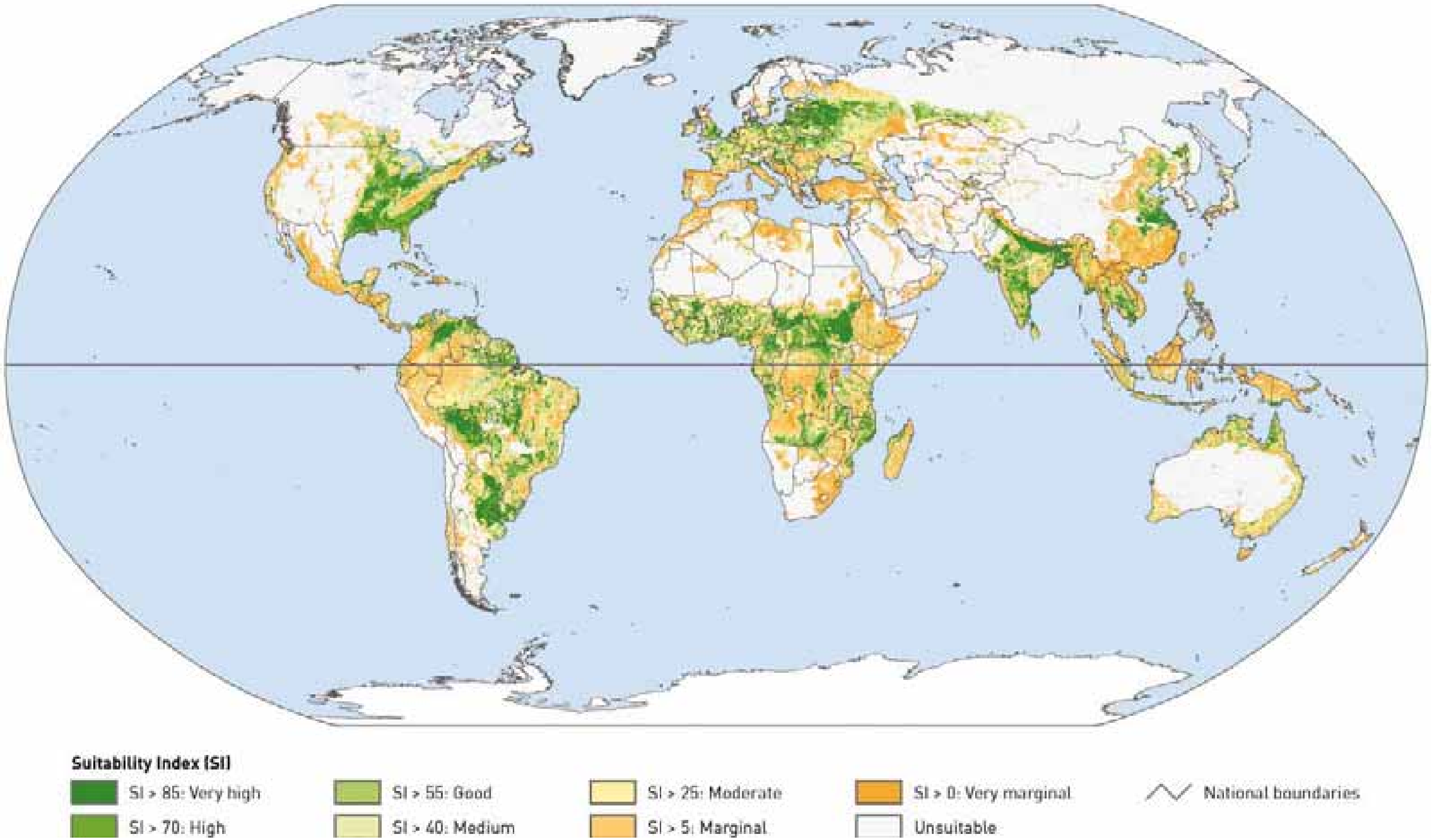
Source: LEAD. The fraction of total production dedicated to feed was estimated at national level (FAO, 2006b) and the ratios applied to the crop production map (You *et al.*, 2006).

Map 10 Current dominant land-use in areas with high suitability for pasture but no current use as pasture



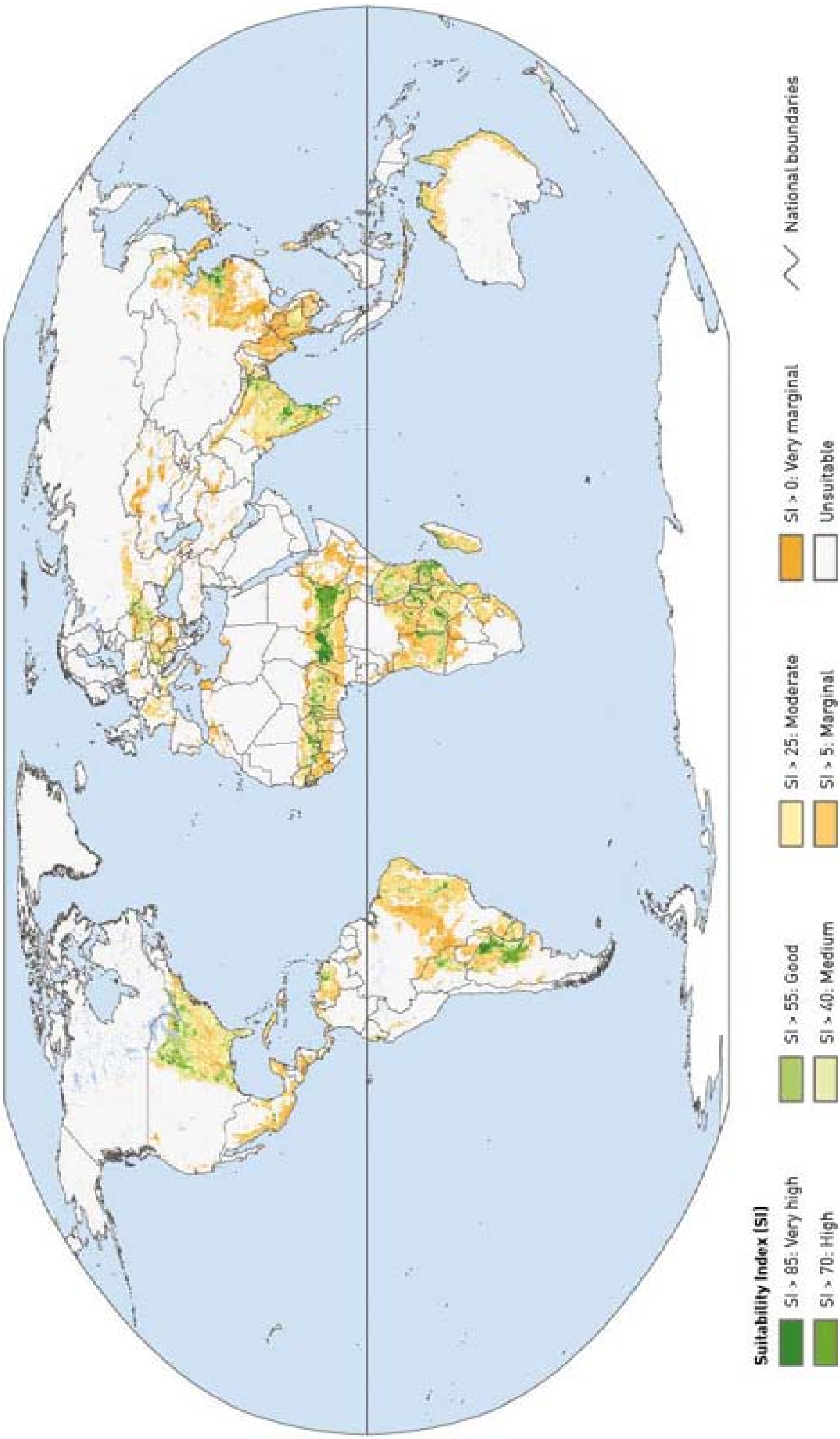
Source: LEAD. Current dominant land-use is displayed for cells with estimated high suitability for pasture (FAO, 2006f) and less than one-third of the area dedicated to pasture (FAO, 2006f).

Map 11 Estimated suitability for rainfed cereal production – high level of input



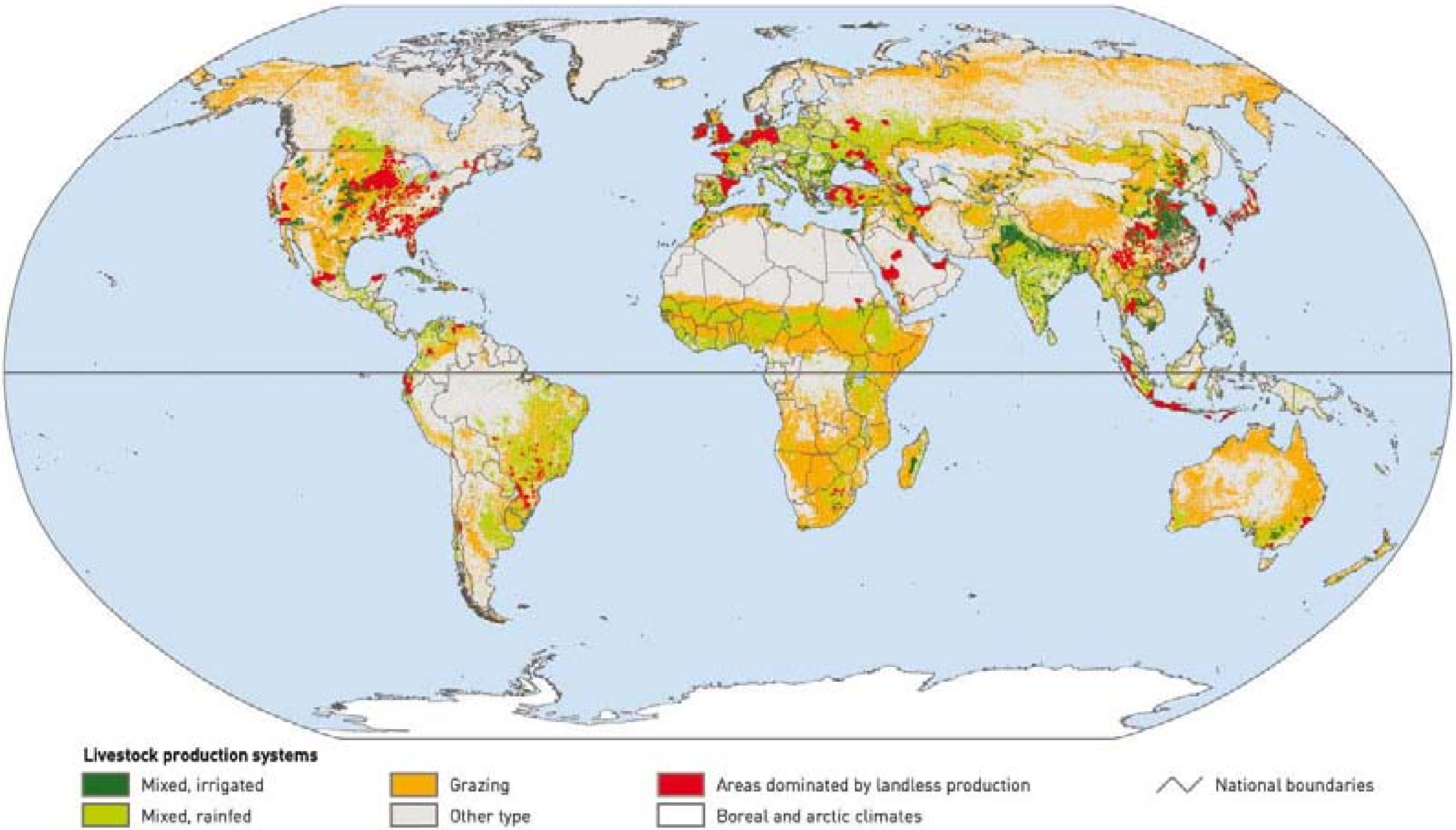
Source: FAO, 2000b.

Map 12 Estimated suitability for soybean production – maximising technology



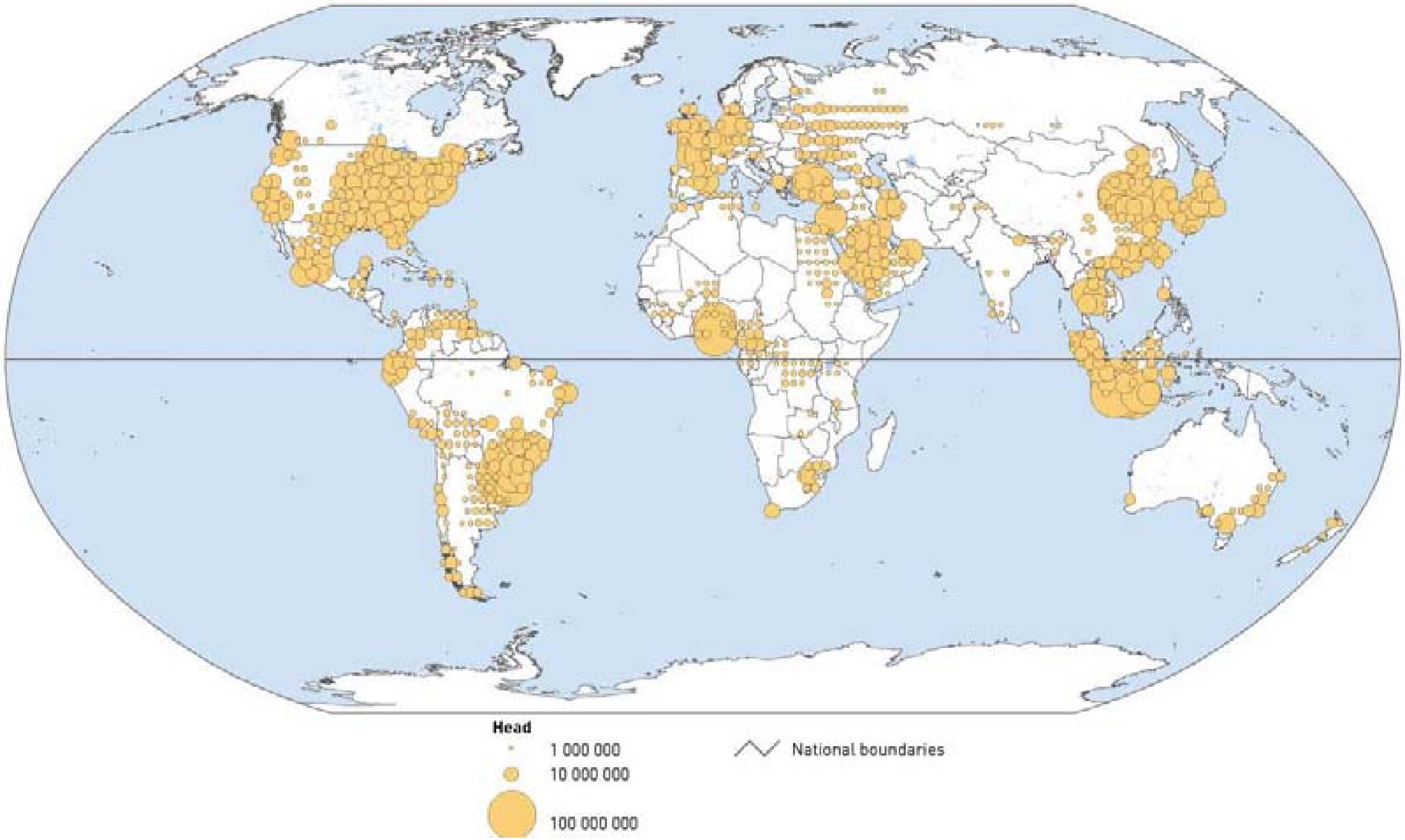
Source: FAO, 2000b.

Map 13 Estimated distribution of livestock production systems



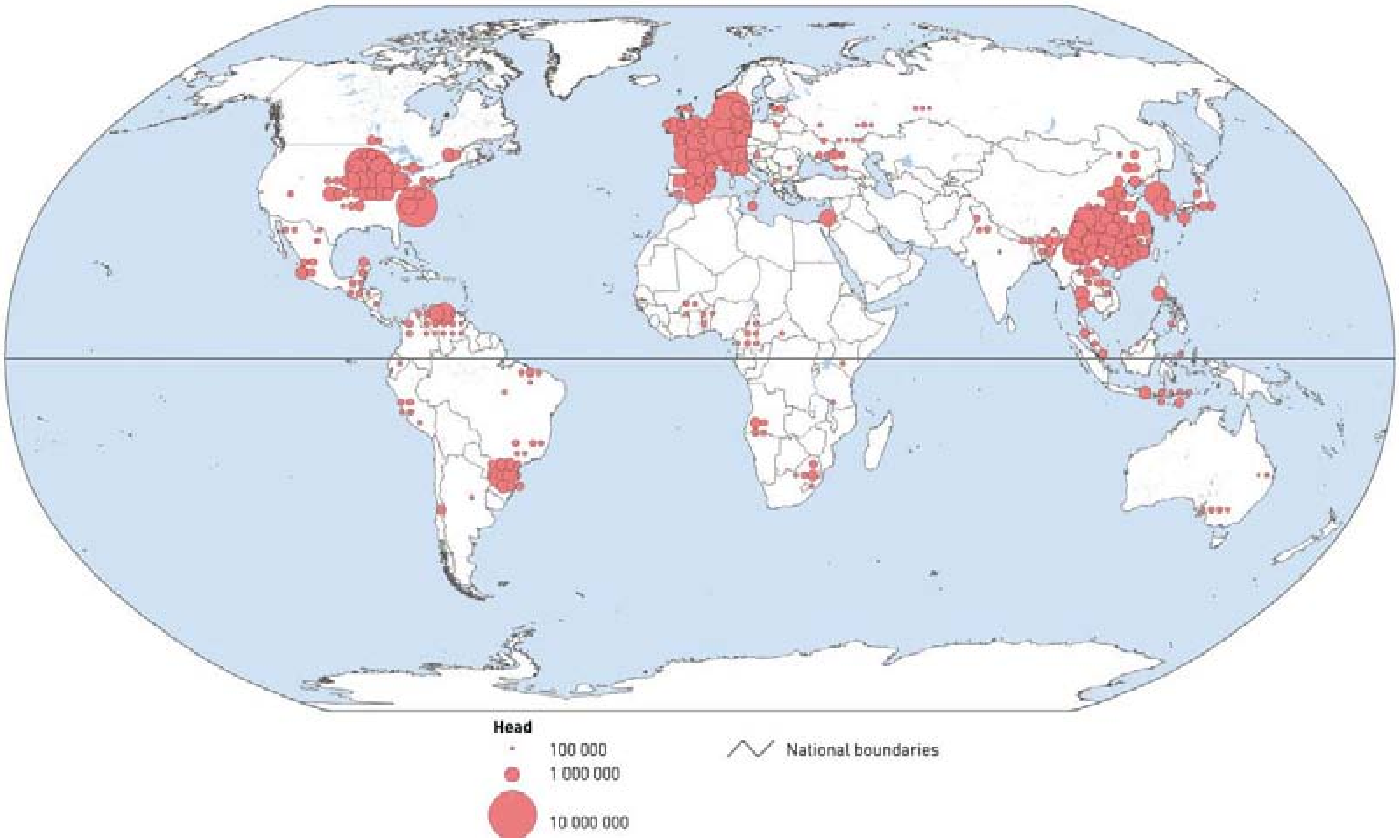
Land Cover 2000, available at www-gvm.jrc.it/glc2000/) and irrigated areas (Global Map of Irrigated Areas. Version 2.1, Siebert *et al.*, 2001). Industrial (landless) production dominated areas refers exclusively to monogastric production. Land-based system held pig and poultry populations are estimated locally according to the approach of Gilbert *et al.* (2004), using total local animal population data (see Maps 16 and 17), national level land-based production estimates (Groenewold, 2004), national human agricultural populations (FAO, 2006b) and a global rural population density grid (LandScan, 2003). Areas dominated by industrial production systems are sub-national administrative areas in which the aggregated land based system populations produce less than half of the areas total production, accounting for the higher productivity of industrial systems.

Map 14 **Estimated distribution of industrially produced poultry populations**



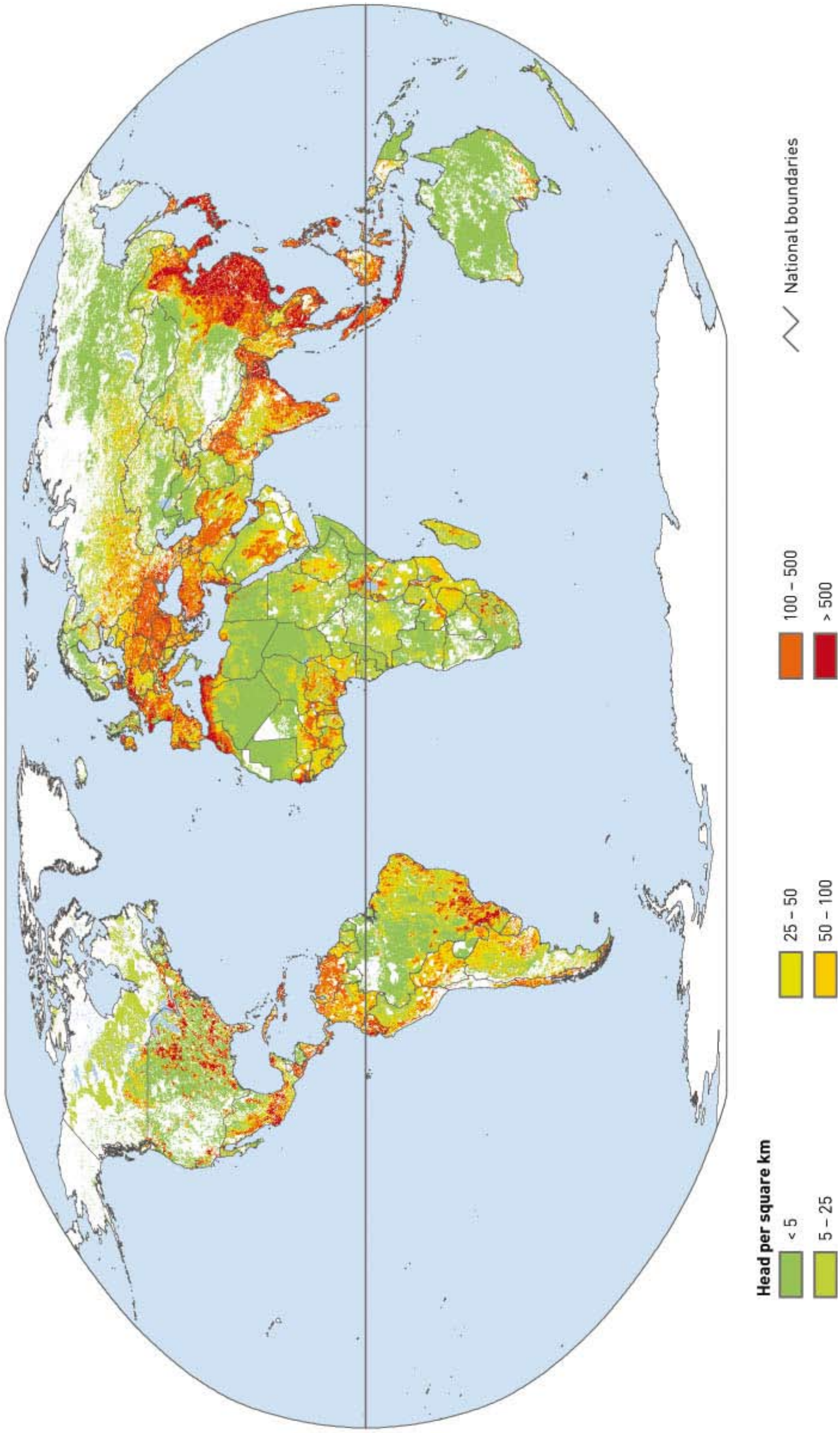
Source: LEAD. The industrial poultry populations result from the difference between the local total population and the locally estimated land-based system held population (see Map 13). Only the sub-national areas where industrial production is dominant are considered (see Map 13). The corresponding industrial poultry populations have been aggregated within each cell of a global grid of 250 x 250 km cell size (cylindrical equal area projection, here re-projected to Robinson). Only industrial populations per cell of over 1 million heads are represented.

Map 15 **Estimated distribution of industrially produced pig populations**



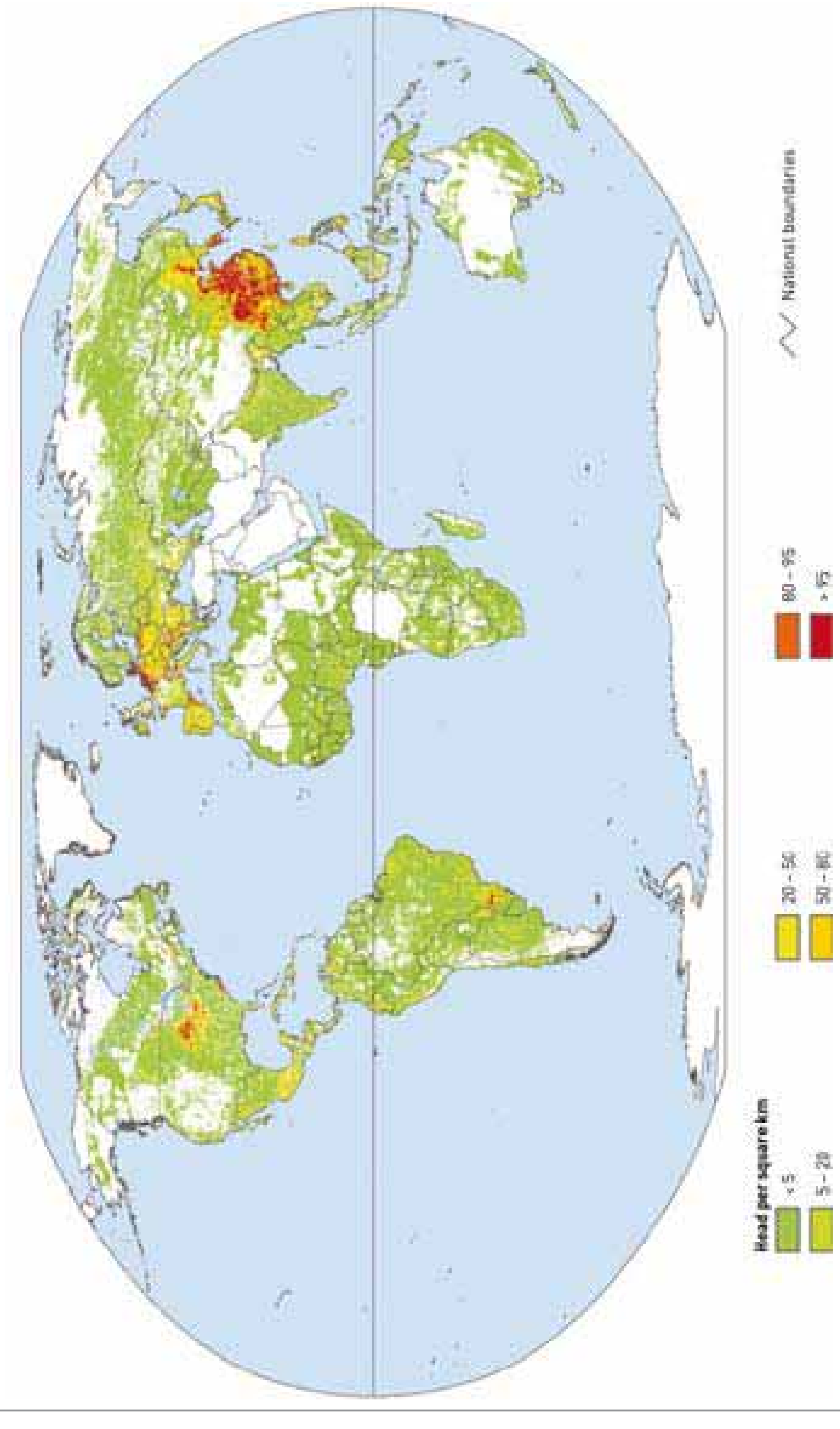
Source: LEAD. The industrial poultry populations result from the difference between the local total population and the locally estimated land-based system held population (see Map 13). Only the sub-national areas where industrial production is dominant are considered (see Map 13). The corresponding industrial poultry populations have been aggregated within each cell of a global grid of 250 x 250 km cell size (cylindrical equal area projection, here re-projected to Robinson). Only industrial populations per cell of over 1 million heads are represented.

Map 16 **Estimated distribution of poultry**



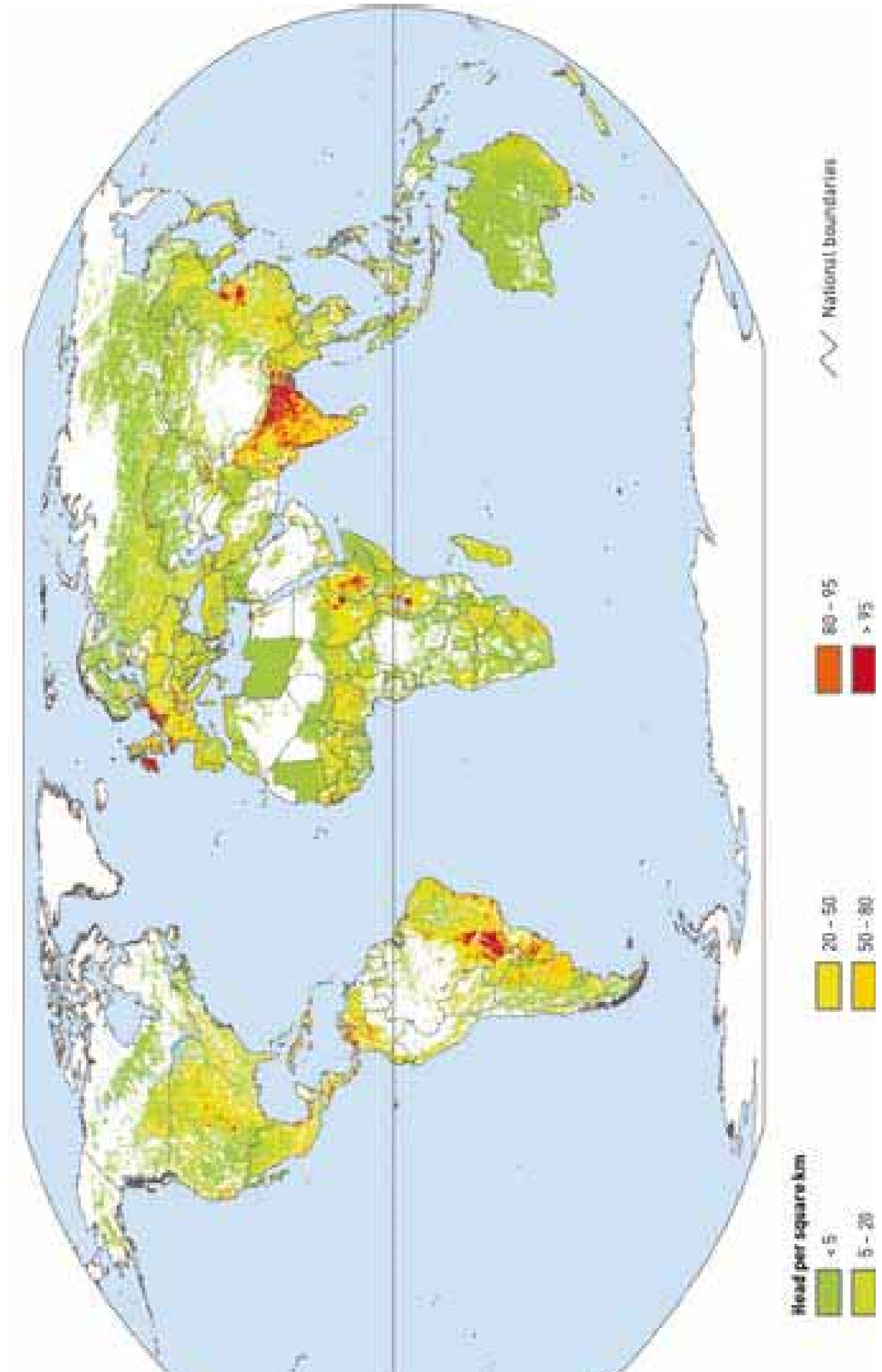
Source: FAO, 2006g.

Map 17 Estimated distribution of pigs



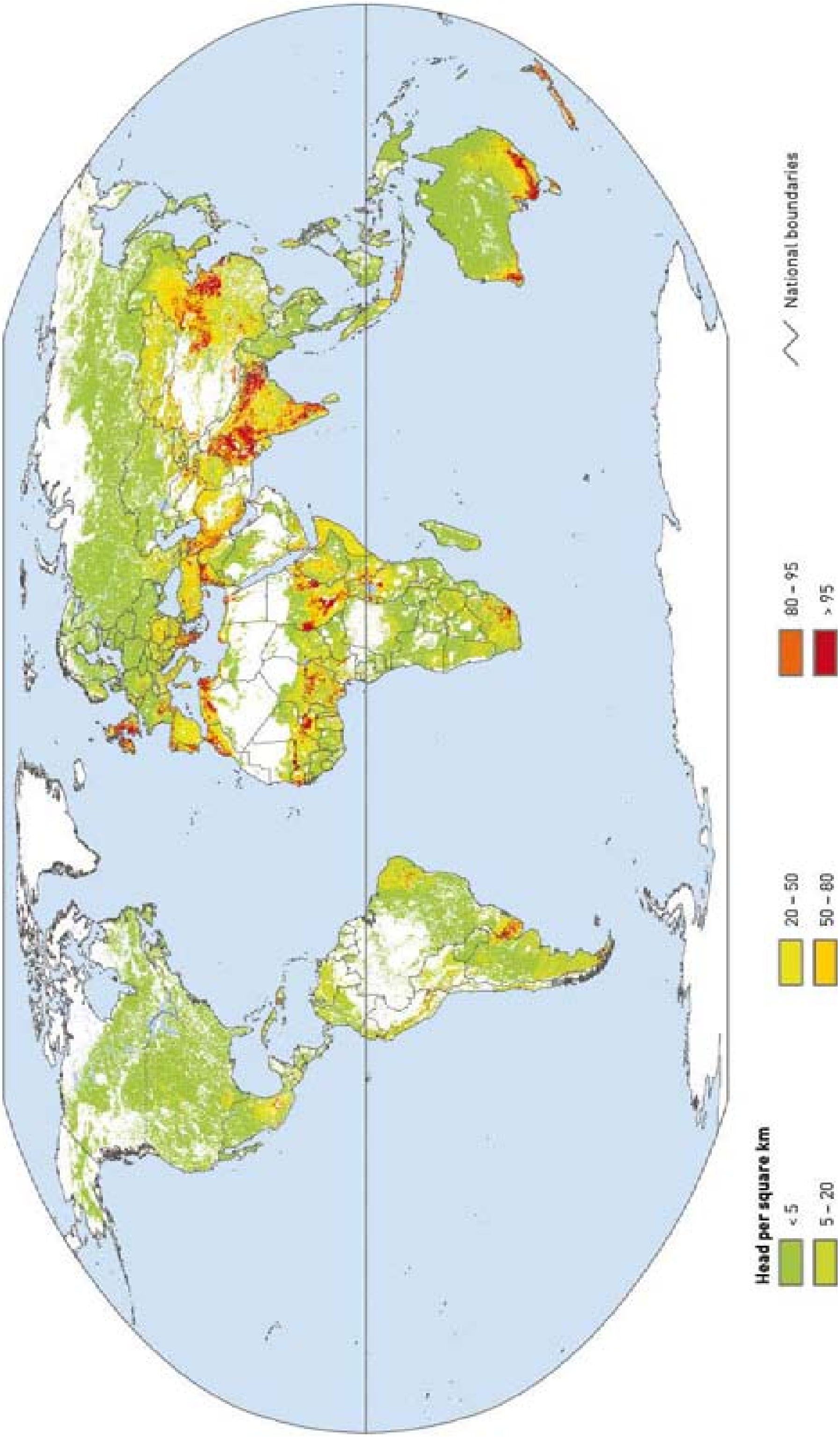
Source: FAO, 2006g.

Map 18 Estimated distribution of cattle



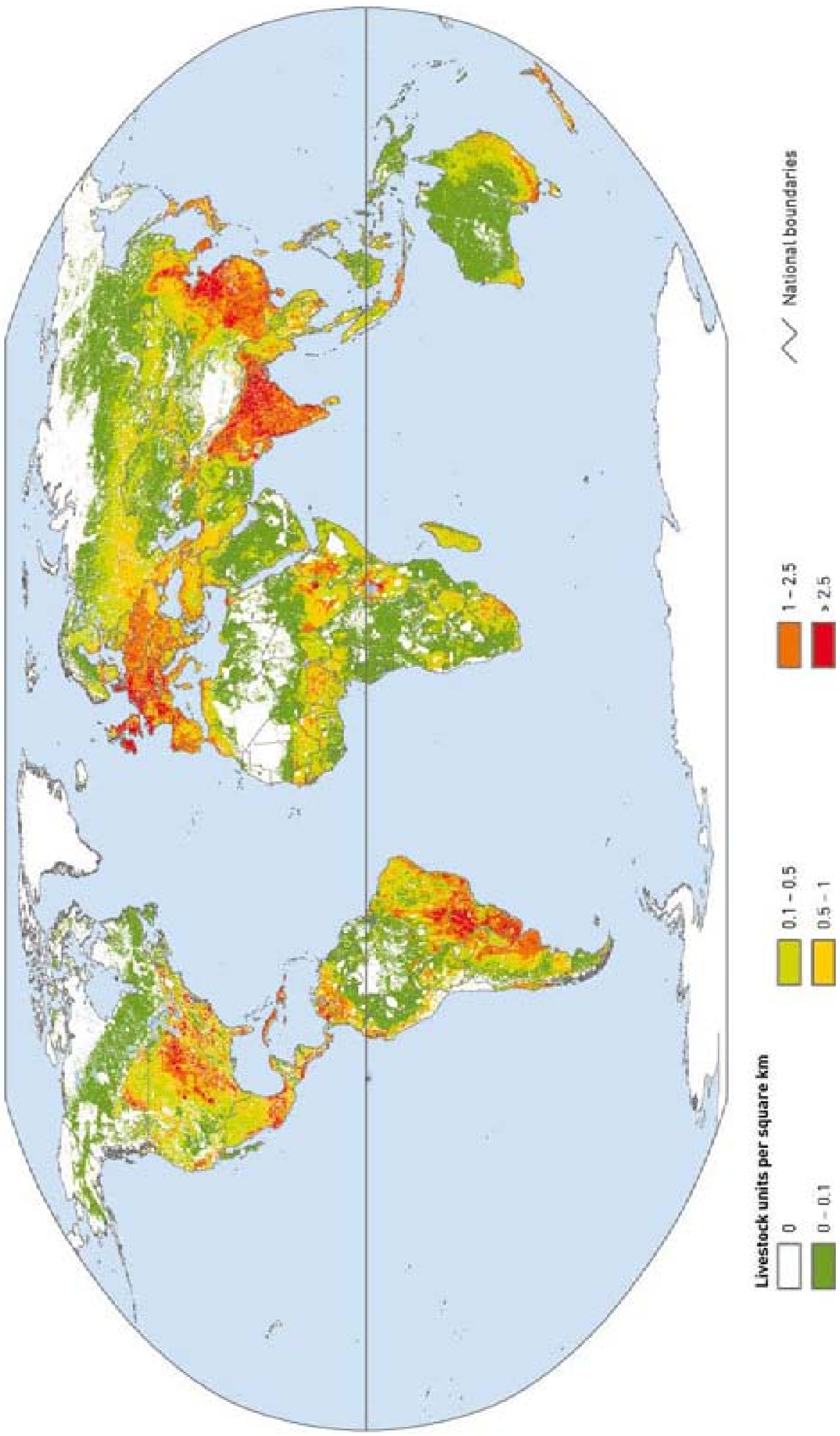
Source: FAO, 2006g.

Map 19 Estimated distribution of small ruminants



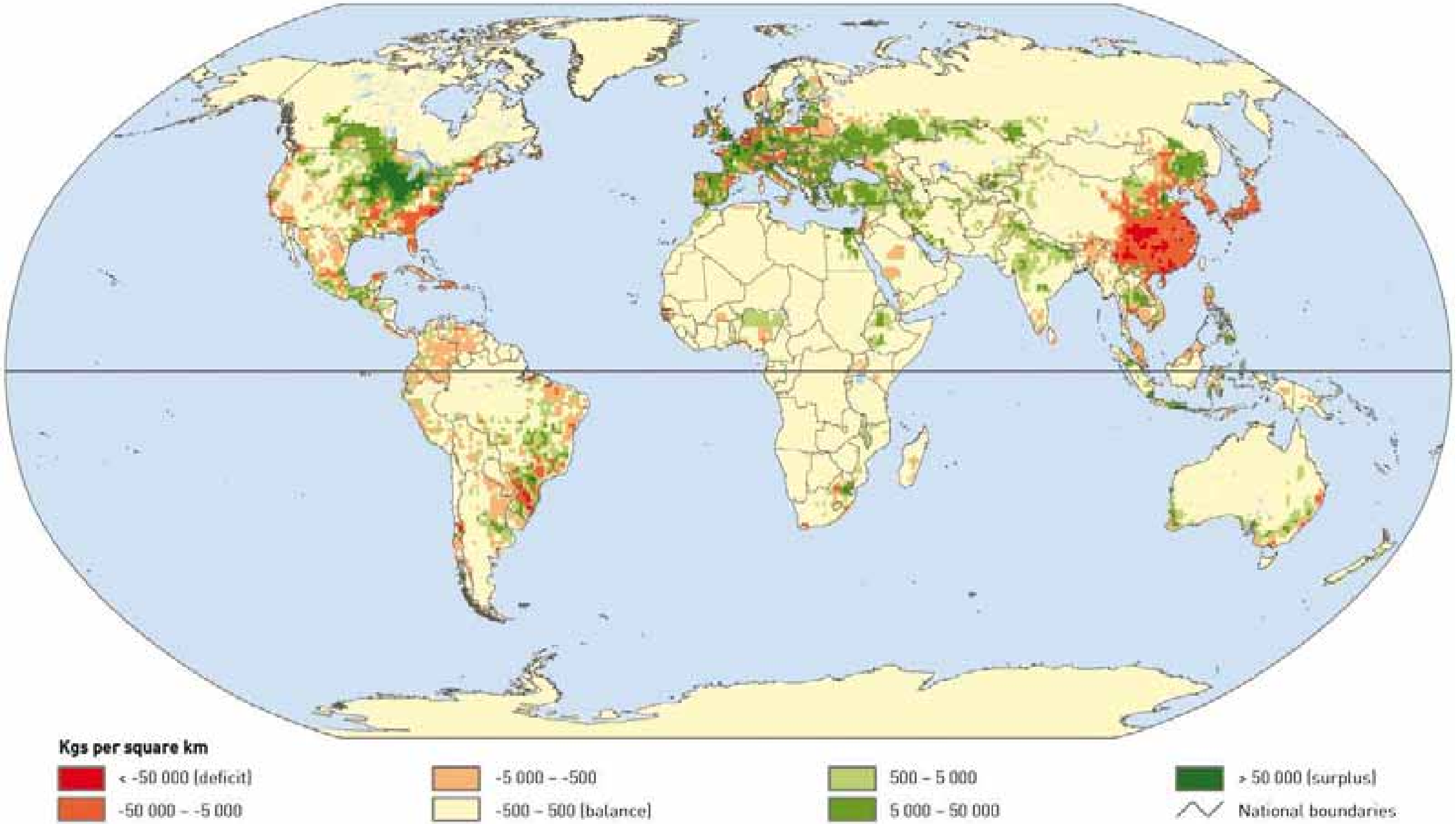
Source: FAO, 2006g.

Map 20 Estimated aggregated distribution of pigs, poultry, cattle and small ruminants



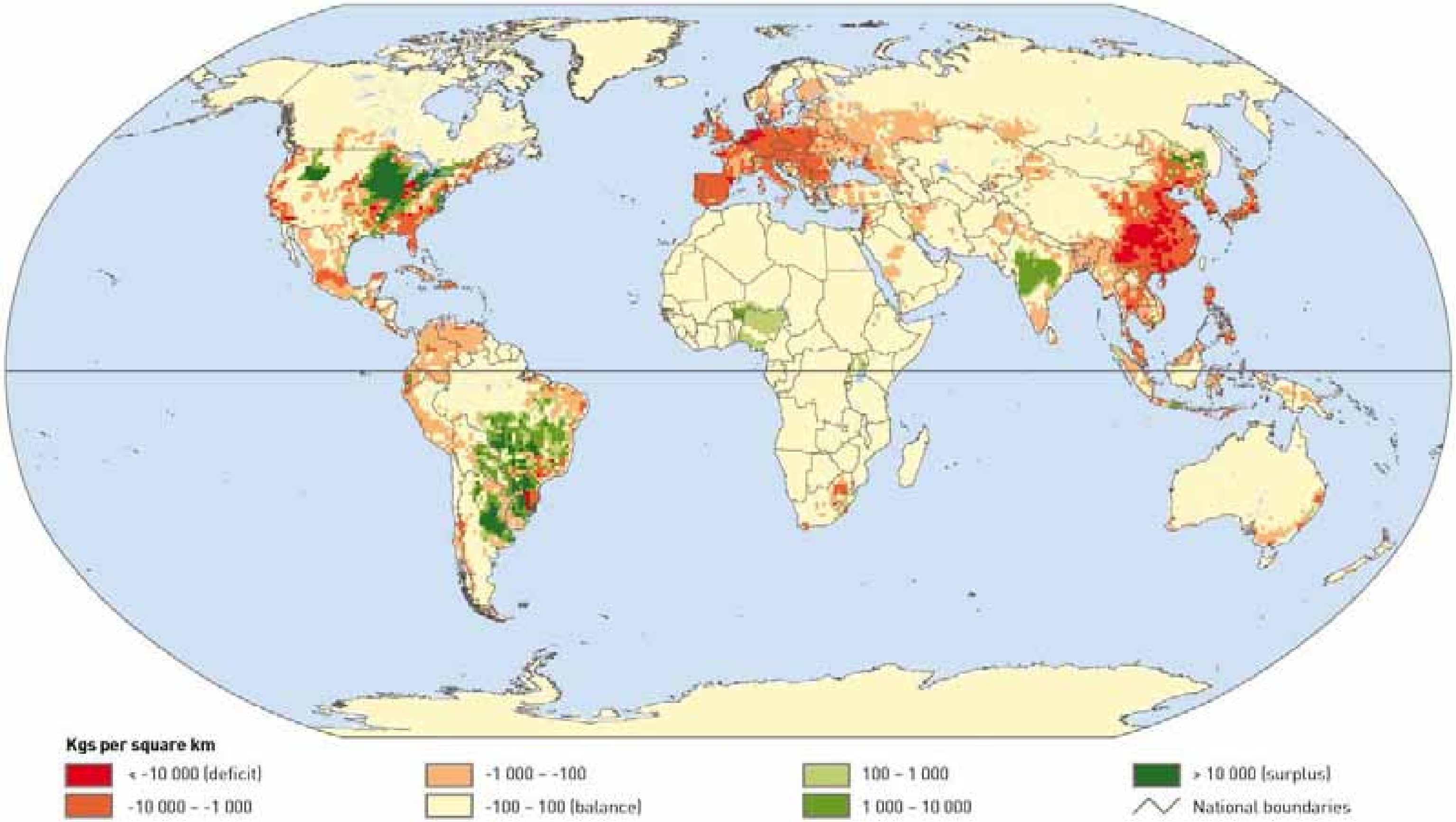
Source: FAO, 2006g.

Map 21 Estimated feed surplus/deficit – cereals (pig and poultry)



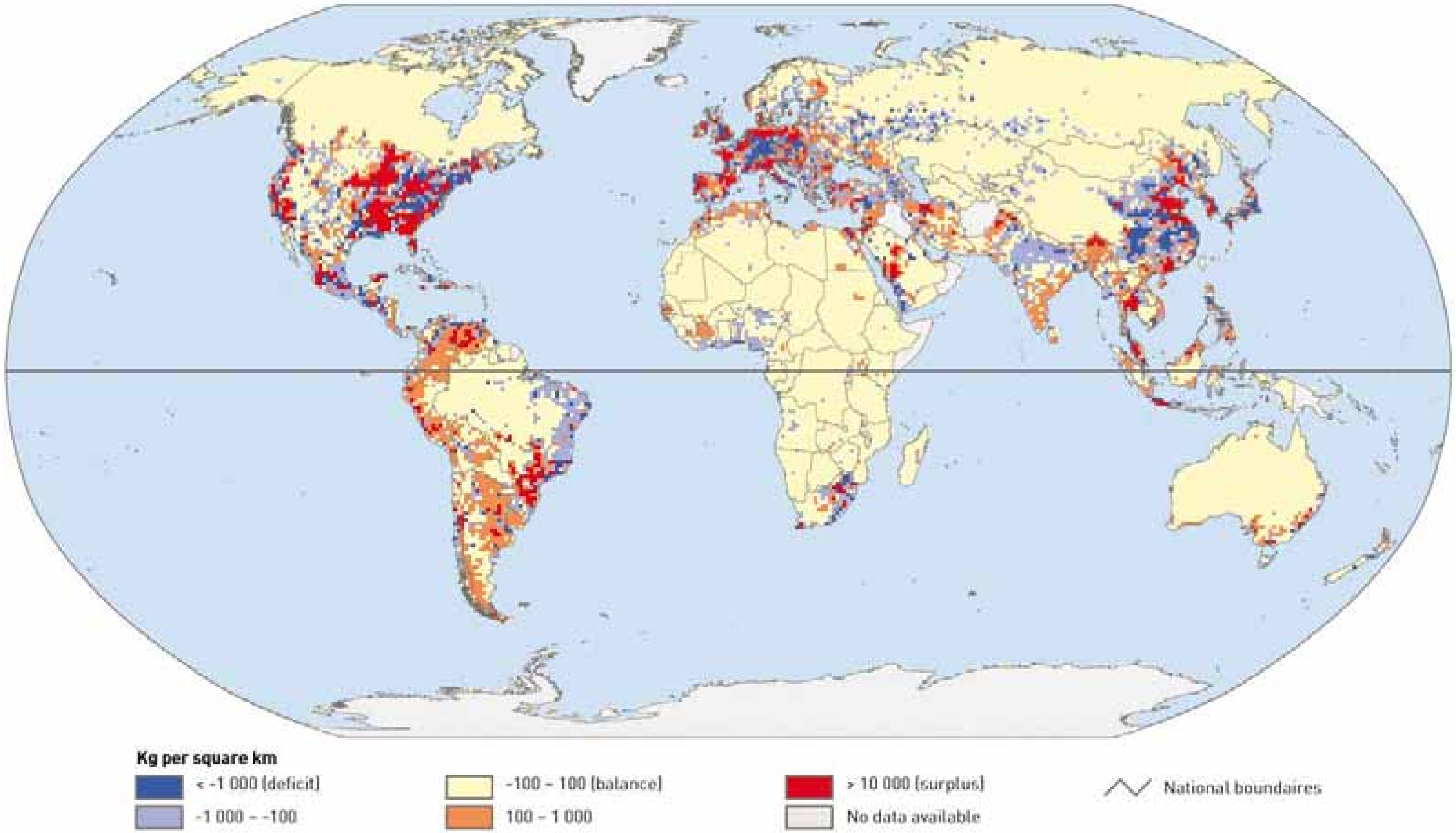
Source: LEAD. For each 100 x 100 km cell, the balance is calculated as the difference between the estimated cumulated maize, wheat and barley (MWB) production for pig and poultry feed and the MWB consumption by pig and poultry. The production map is derived from the estimated MWB production for animal feed (Map 8), removing the fraction of MWB feed dedicated to ruminants (Galloway *et al.*, 2006). The consumption map was calculated from the pig and poultry meat production maps (see source of Maps 23 and 24). National level indexes derived were first used to estimate live weight production and total feed consumption (FAO, 2006b). For each country, the share of MWB in the feed basket composition was then extrapolated from available data (e.g. Chapter 2, Figures. 2.6 and 2.7). This share was finally used to calculate the MWB consumption by pig and poultry in each cell.

Map 22 Estimated feed surplus/deficit – soymeal (pig and poultry)



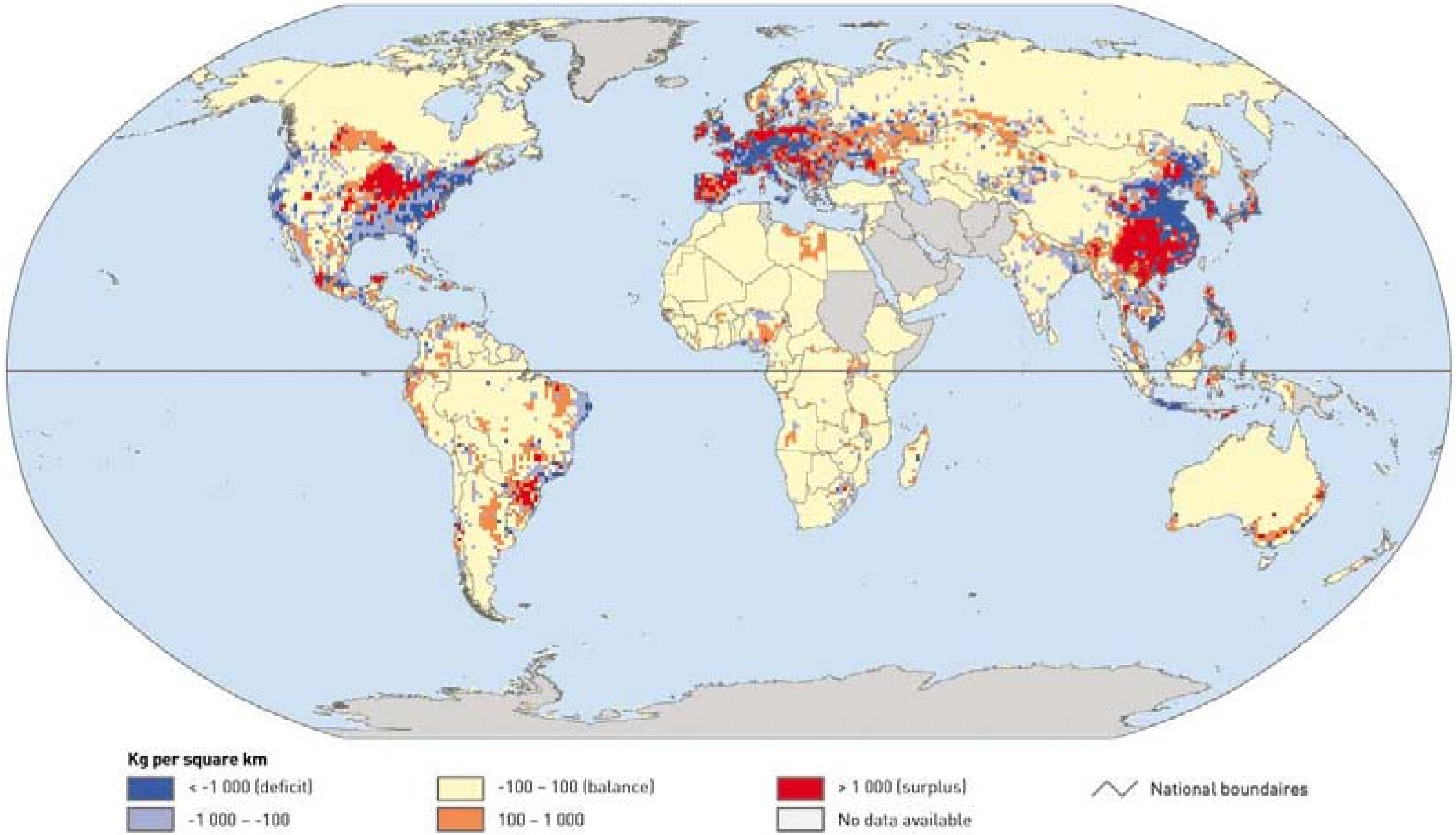
Source: LEAD. For each 100 x 100 km cell, the balance is calculated as the difference between the estimated soymeal production for pig and poultry feed and the soymeal consumption by pig and poultry. The soymeal production map is derived from the estimated soybean production for animal feed (map 8), removing the fraction dedicated to ruminants (Galloway *et al.*, 2006) and applying a bean to meal weight conversion factor (Schnittker, 1997). The consumption map was calculated from the pig and poultry meat production maps (see source of Maps 23 and 24). National level indexes derived were first used to estimate live weight production and total feed consumption (FAO, 2006b). For each country, the share of soymeal in the feed basket composition was then extrapolated from available data (e.g. Chapter 2, Figures. 2.6 and 2.7). This share was finally used to calculate the soymeal consumption by pig and poultry in each cell.

Map 23 **Estimated poultry meat surplus/deficit**



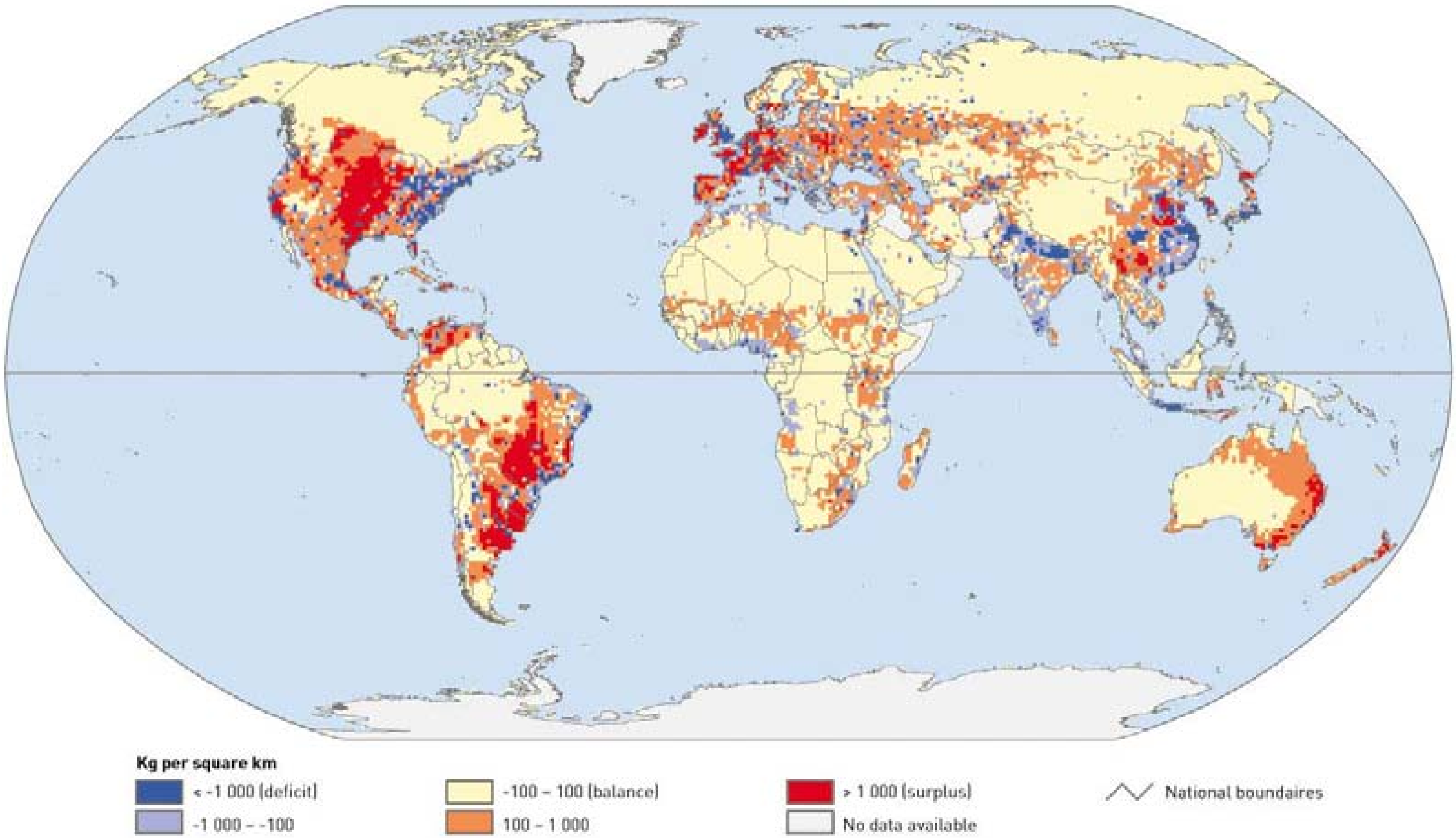
Source: LEAD – For each 100 x 100 km cell, the balance is calculated as the difference between estimated poultry meat production and consumption. The production map is based on national level statistics (FAO, 2006b) distributed along animal densities (see map 16) corrected for the level of production intensity (see map 14). The consumption map was calculated by distributing national level statistics (FAO, 2006b) along human population (see map 3). In developing countries, higher consumption levels were attributed to urban areas (LandScan, 2005) than to rural areas.

Map 24 Estimated pig meat surplus/deficit



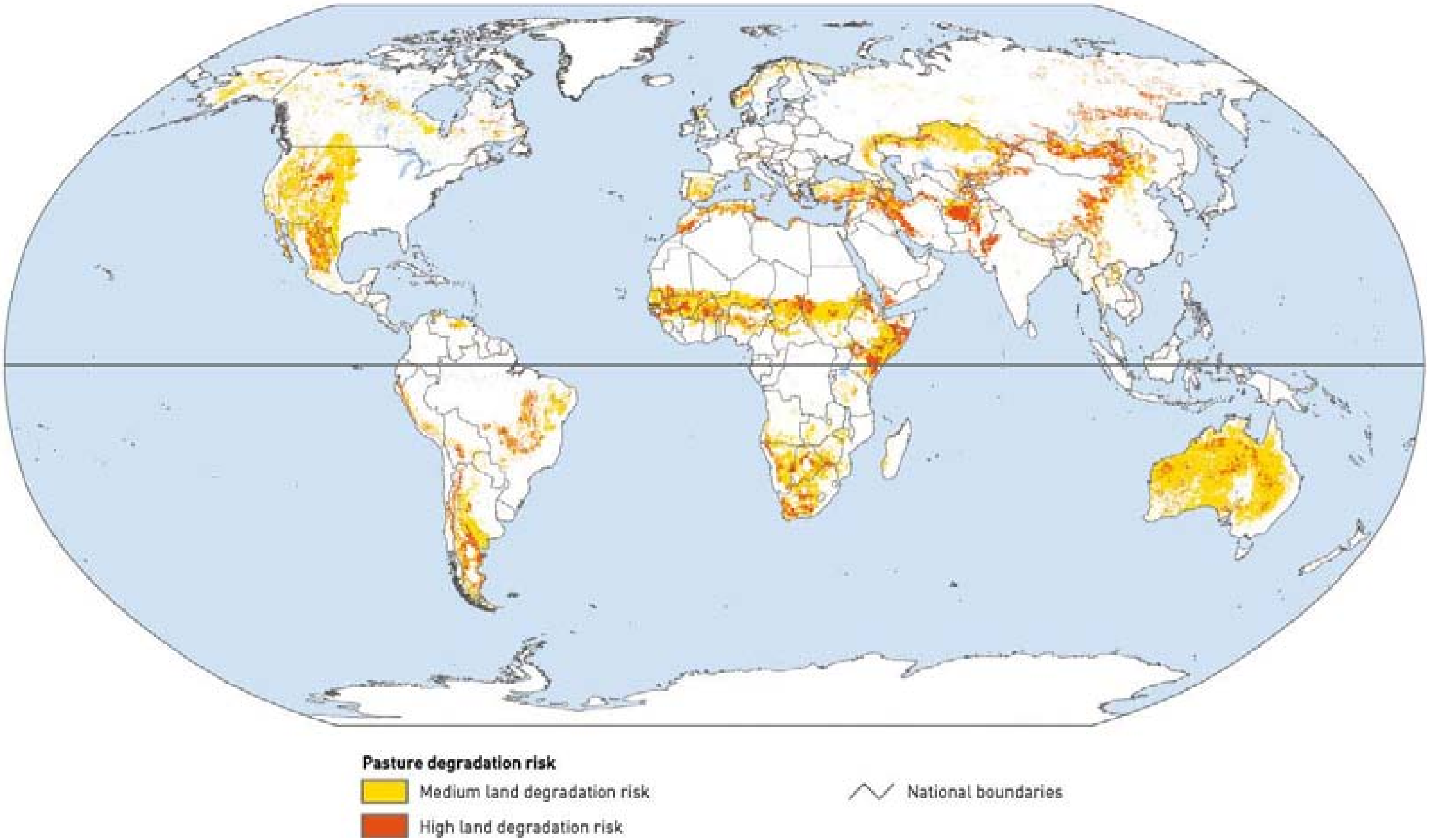
Source: LEAD. For each 100x100 km cell, the balance is calculated as the difference between estimated pig meat production and consumption. The production map is based on national level statistics (FAO, 2006b) distributed along animal densities (see map 17) corrected for the level of production intensity (see map 15). The consumption map was calculated by distributing national level statistics (FAO, 2006b) along human population (see map 3). In developing countries, higher consumption levels were attributed to urban areas (LandScan, 2005) than to rural areas.

Map 25 **Estimated beef surplus/deficit**



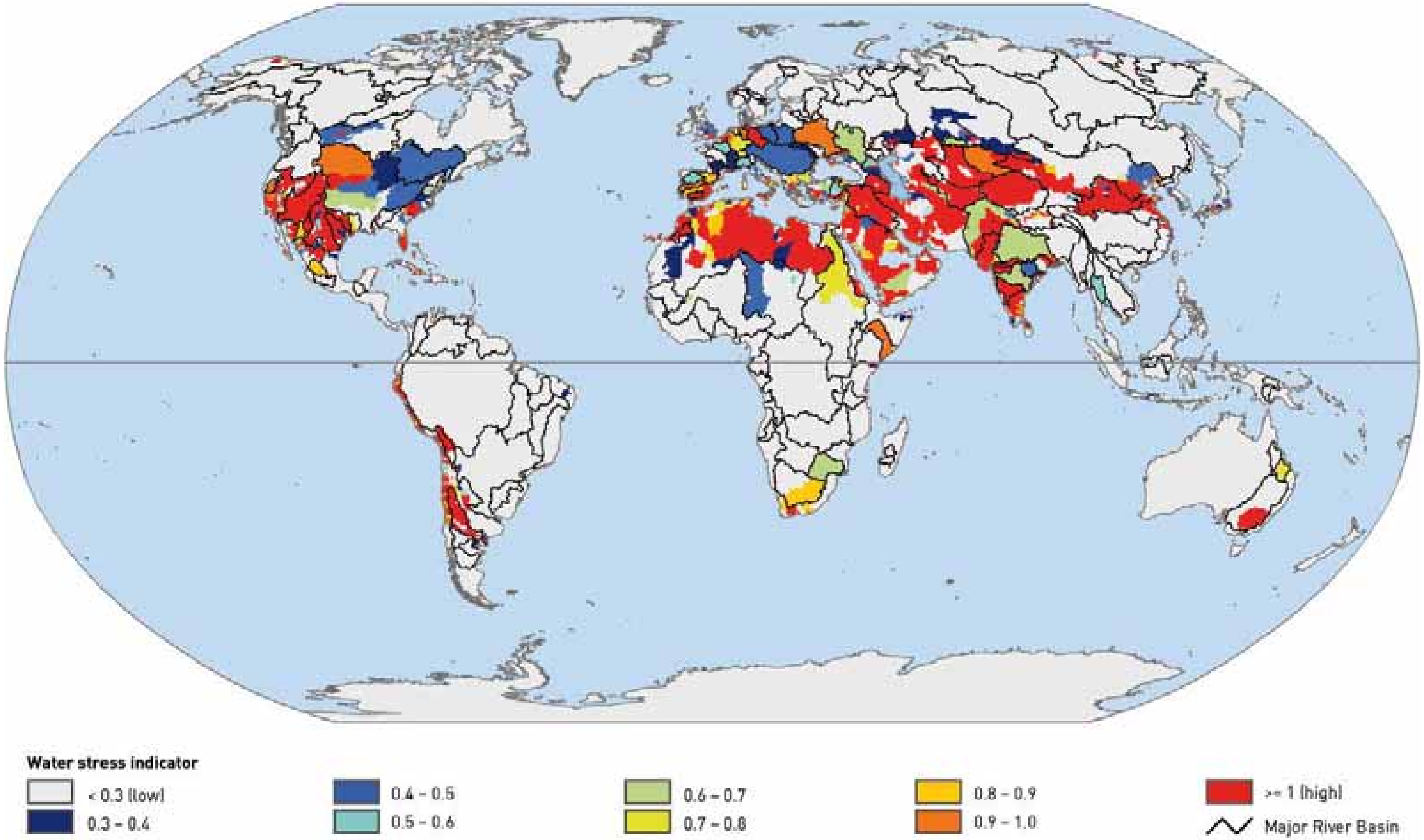
Source: LEAD. For each cell, the balance is calculated as the difference between estimated beef production and consumption. The production map is based on national level statistics (FAO, 2006b) distributed along animal densities (see map 18). The consumption map was calculated by distributing national level statistics (FAO, 2006b) along human population (see map 3). In developing countries, higher consumption levels were attributed to urban areas (LandScan, 2005) than to rural areas.

Map 26 Pasture degradation risk in the dry and cold lands



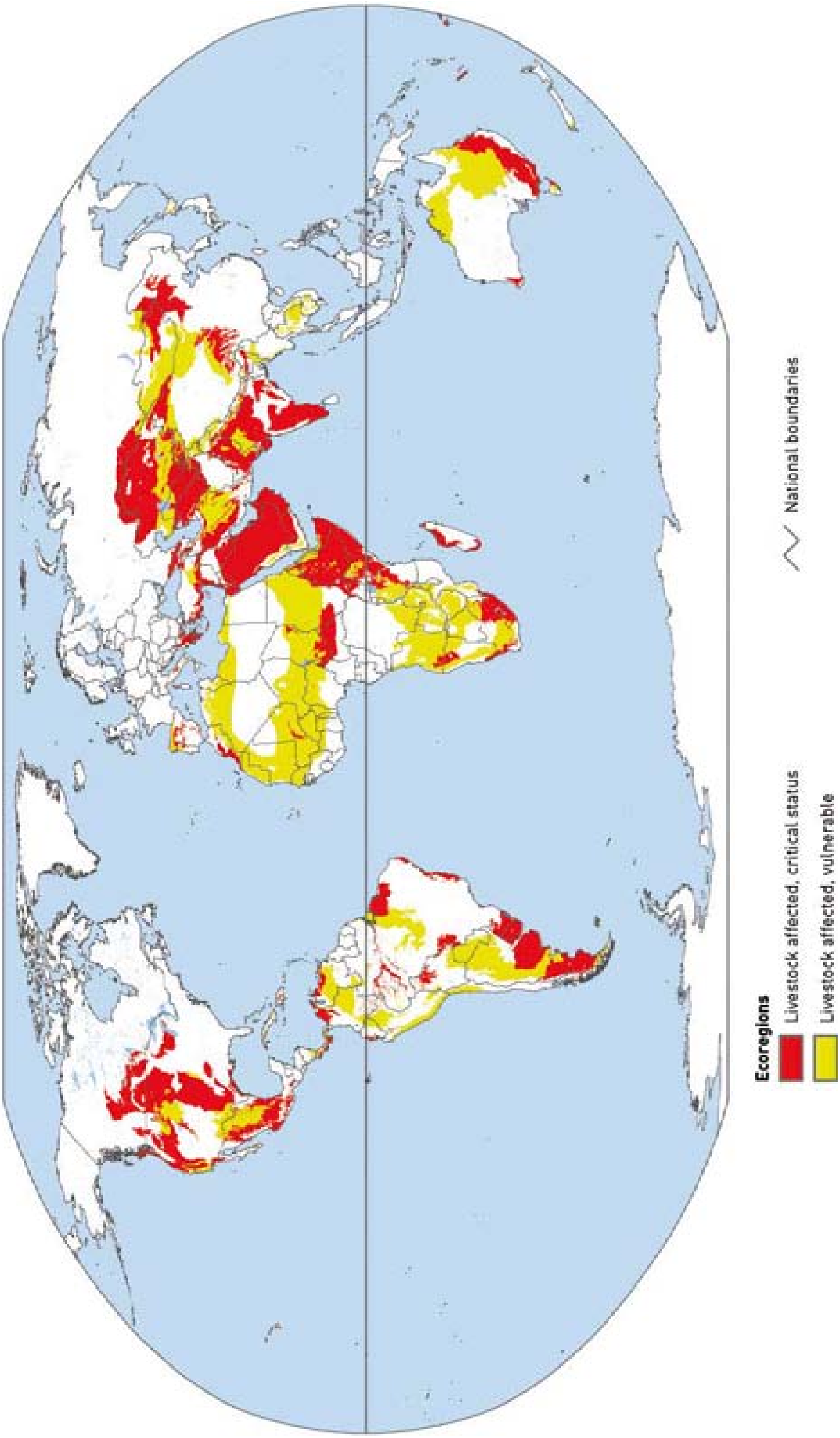
Source: LEAD. Cells with a low or very low suitability for pasture (FAO, 2006f) and at least a third of the area used as pasture (FAO, 2006f) were assigned to the "high risk" class, whereas cells with a medium suitability for pasture (FAO, 2006f) and at least a third of the area used as pasture (FAO, 2006f) were assigned to the "medium risk" class. Both the "high" and "medium" degradation risk classes are assigned to cells with cattle population (see Map 18).

Map 27 Human infringement on environmental water demand (water withdrawal as a proportion of water available for human use)



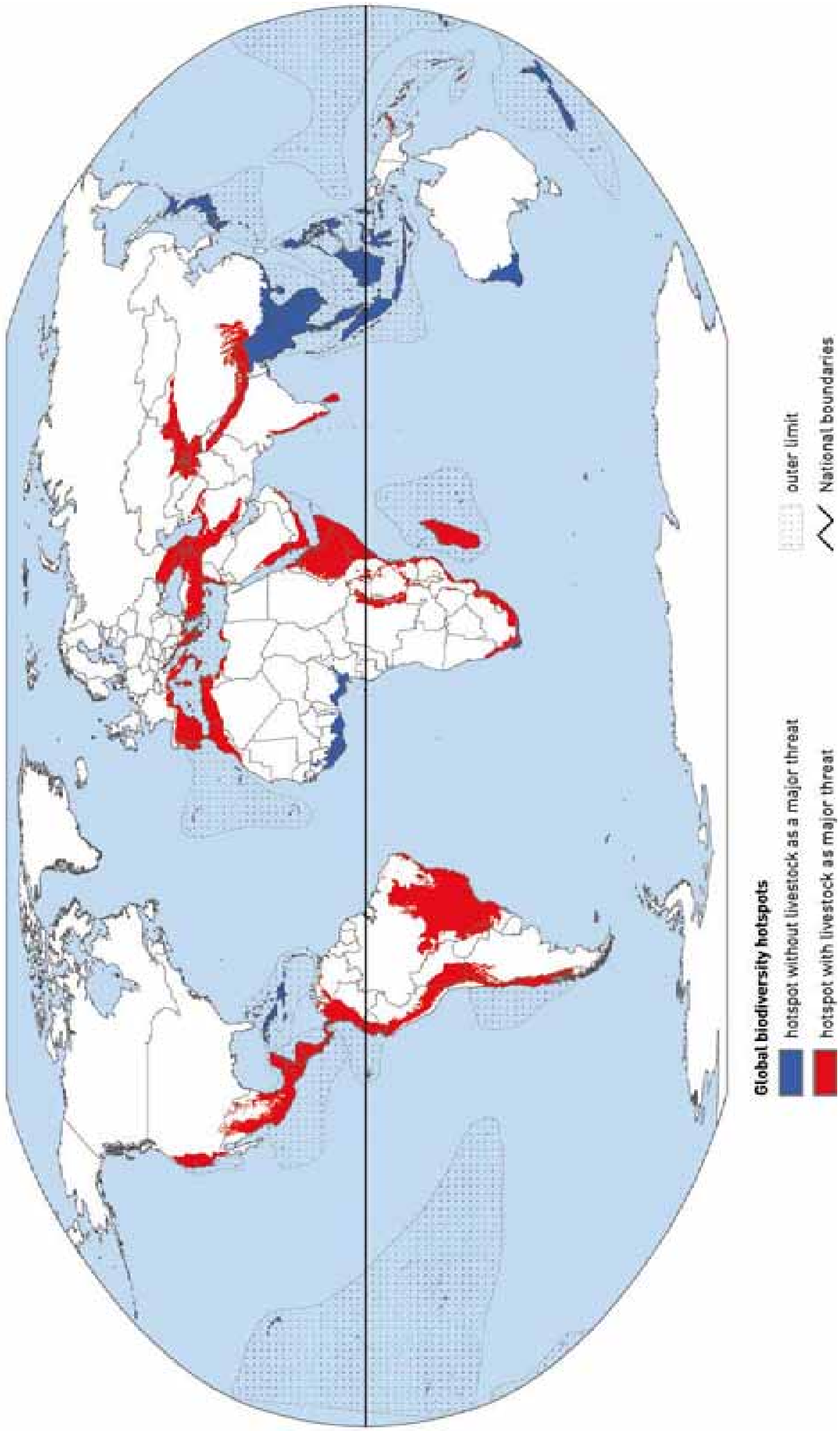
Source: World Resources Institute, 2003.

Map 28 **Ecoregions affected by livestock**



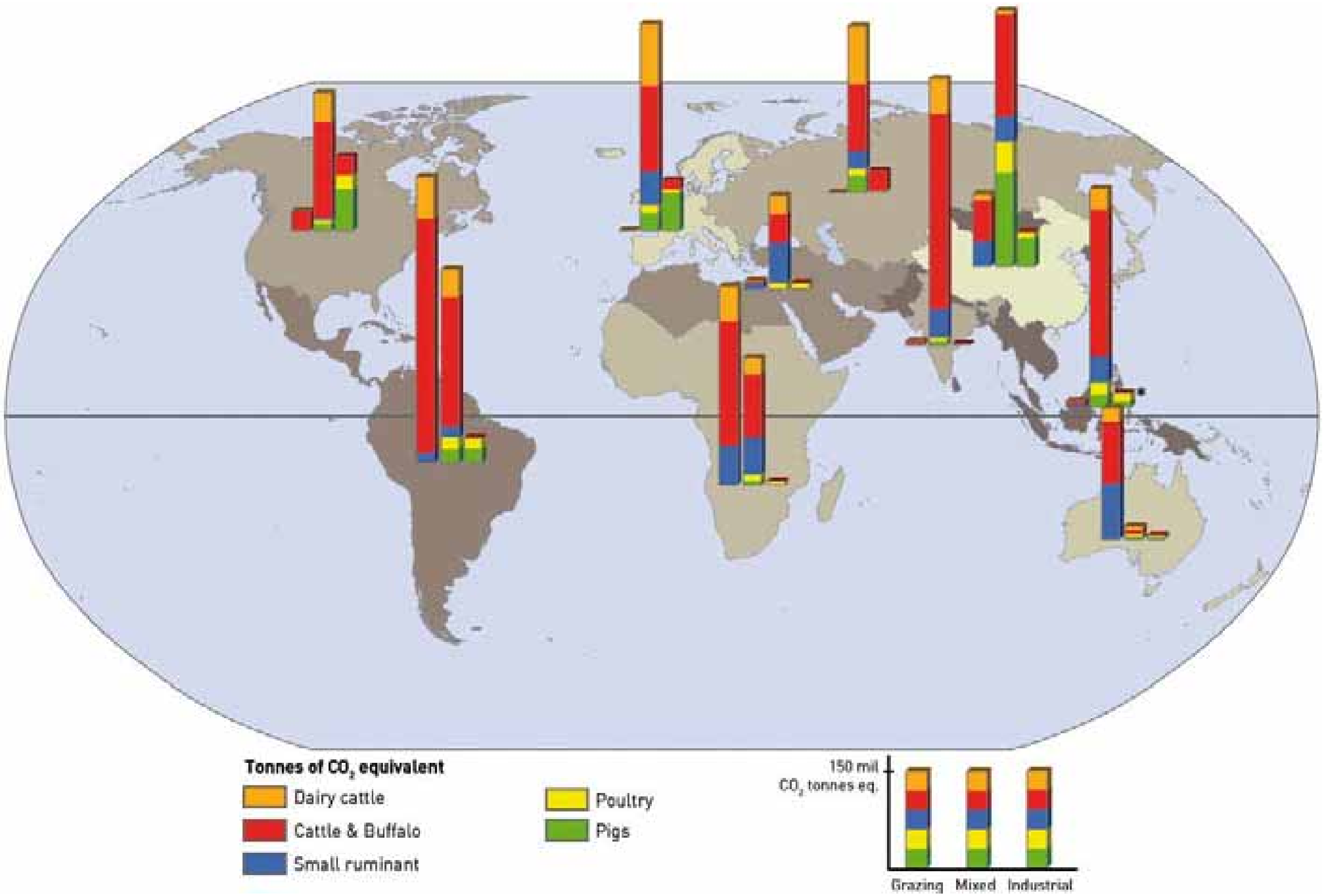
Source: Olson and Dinerstein, 2002.

Map 29 Livestock as an important cause behind global biodiversity hotspots



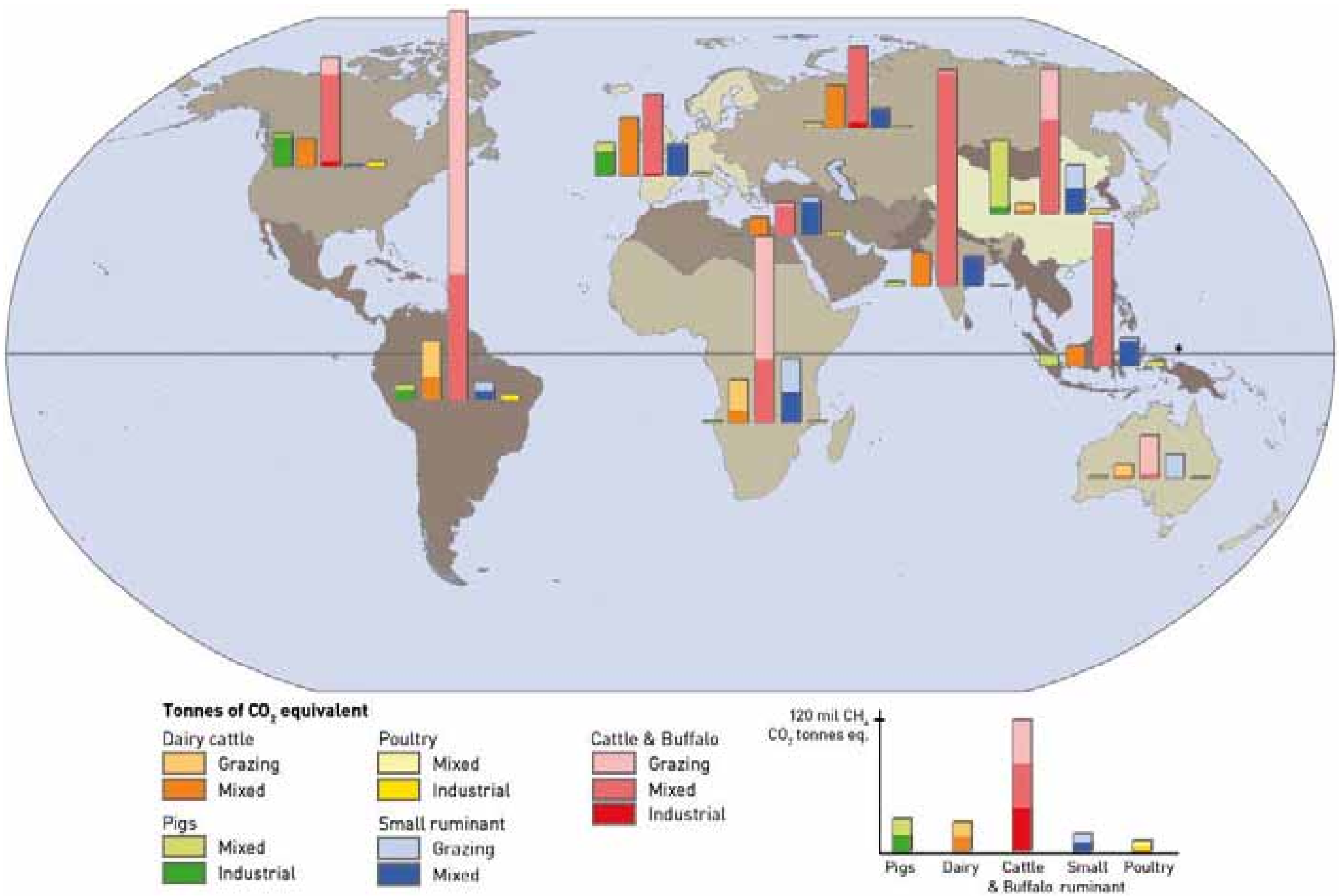
Source: Mittermeier *et al.*, 2004.

Map 30 Total greenhouse gas emissions from enteric fermentation and manure per species and main production system



Source: LEAD. Based on region and production system specific population estimates (Groenewold, 2005) and emission factors (see Chapter 3, Box 3.4 and Annex 3.1 and 3.2).
* South and East Asia excluding China and India.

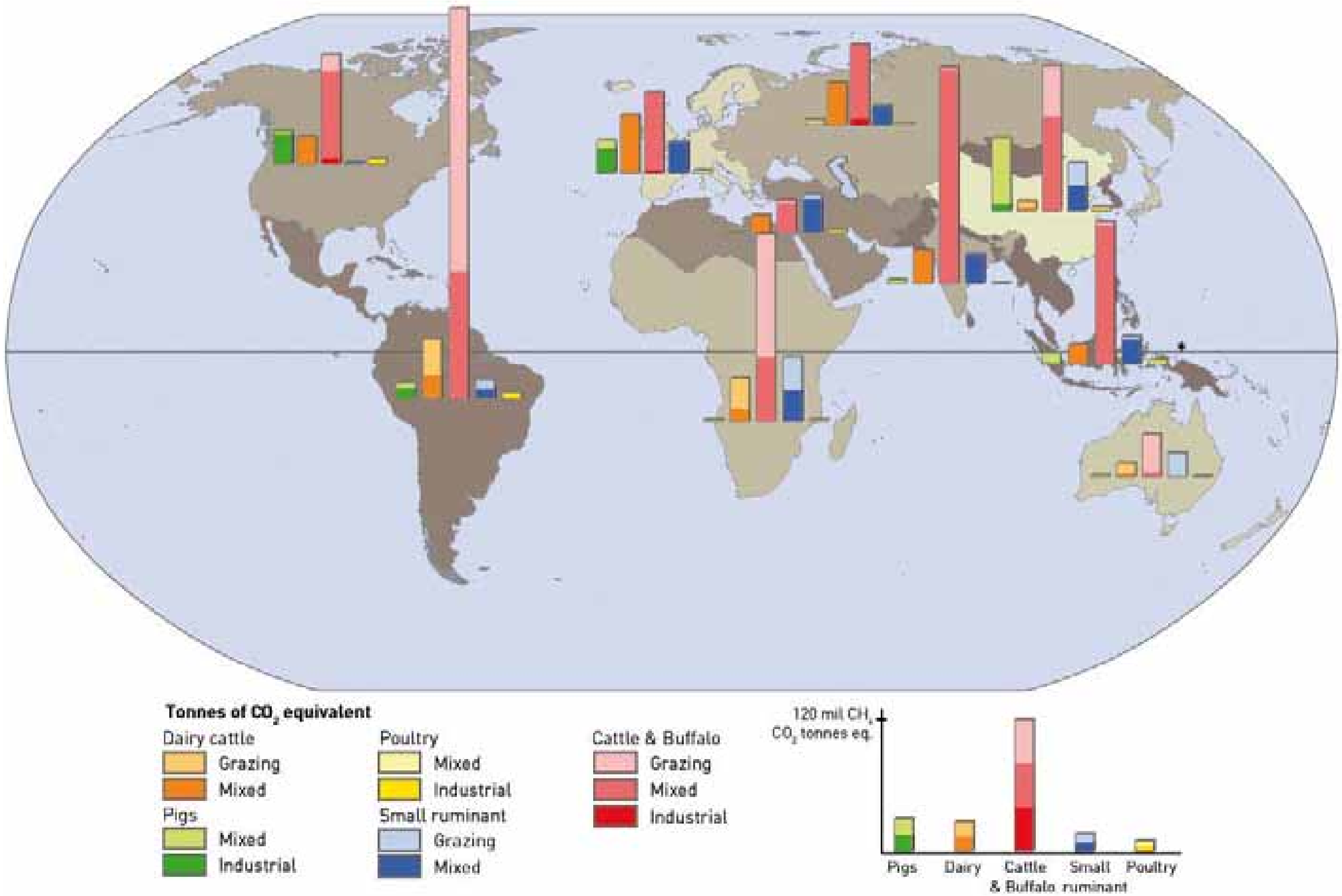
Map 31 **Total methane emissions from enteric fermentation and manure per species and main production system**



Source: LEAD. Based on region and production system specific population estimates (Groenewold, 2005) and emission factors (see Annex 3.1 and 3.2).

* South and East Asia excluding China and India.

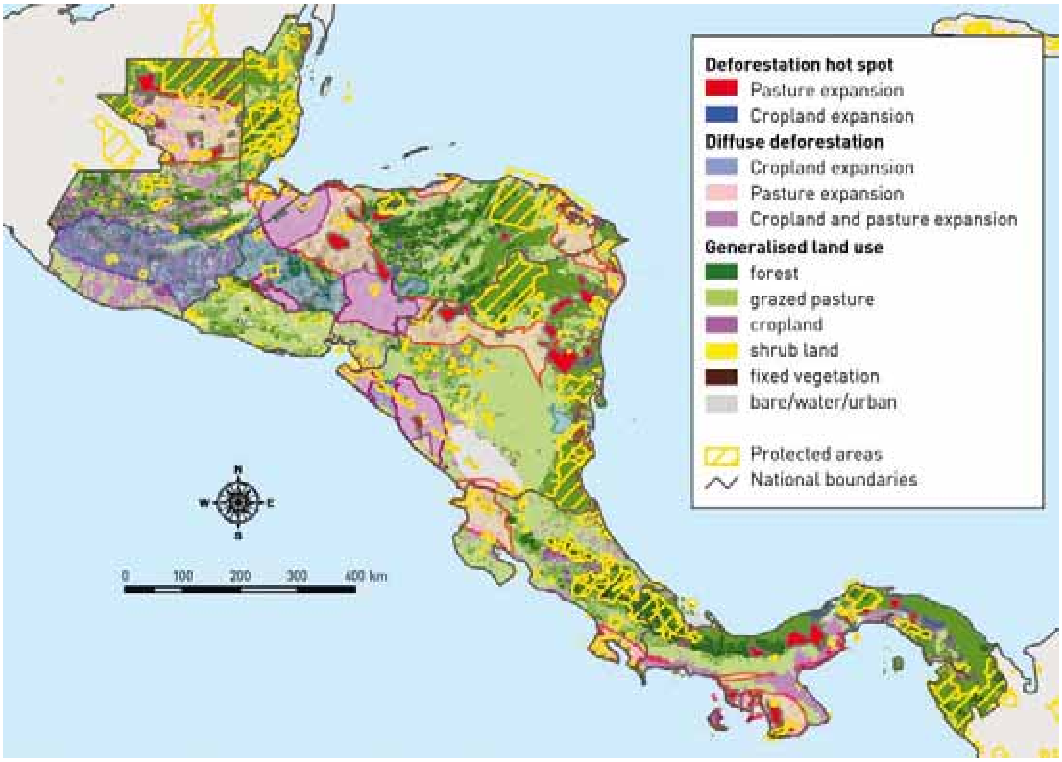
Map 32 **Total nitrous oxide emissions from manure per species and main production system**



Source: LEAD. Based on region and production system specific population estimates (Groenewold, 2005) and emission factors (see Chapter 3, Box 3.4).

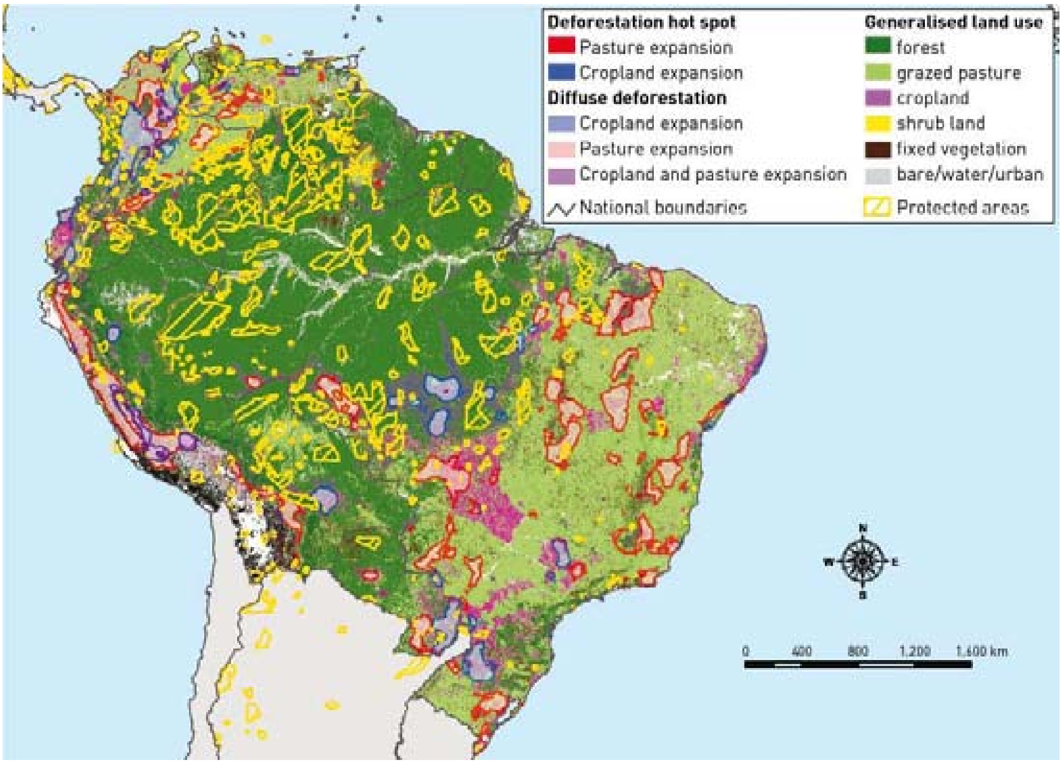
* South and East Asia excluding China and India.

Map 33A Projected expansion of cropland and pasture into Neotropical forest from 2000 to 2010



Source: Wassenaar *et al.*, 2006.

Map 33B Projected expansion of cropland and pasture into Neotropical forest from 2000 to 2010



Source: Wassenaar *et al.*, 2006.

Annex 2

Tables

Annex 2
Tables

Table 1

Regional level trends for three land-use intensification indicators over the period 1961 to 2001

Region	Use of tractors: Annual growth rate (%) 1961–1991	Use of tractors: Annual growth rate (%) 1991–2001	Use of tractors: Crop land[1] per tractor in 2001 (ha)	Use of mineral fertilisers: Annual growth rate (%) 1961–1991	Use of mineral fertilisers: Annual growth rate (%) 1991–2001	Use of mineral fertilisers: Mineral fertilizers used per ha of crop land[1] in 2001 (Kg)	Irrigated area: Annual growth rate (%) 1961–1991	Irrigated area: Annual growth rate (%) 1990–2000	Irrigated area: Share of arable and perm. crops in 2001 (%)
Asia	11.1	1.7	77.3	9.0	1.5	134.7	1.9	1.4	33.5
Oceania	-0.8	-0.9	139.7	0.7	5.6	59.0	2.6	1.8	4.9
Baltic states and CIS	n.d.	n.d.	67.1	n.d.	n.d.	30.2	n.d.	n.d.	49.5
Eastern Europe	7.1	0.2	19.4	1.4	1.2	80.7	3.8	-1.4	10.2
Western Europe	3.1	-0.2	12.0	2.0	-1.5	180.7	1.9	0.9	15.3
North Africa	4.4	1.3	91.8	4.6	2.1	69.5	1.0	1.6	21.7
Sub-Saharan Africa	0.9	-2.8	773.8	5.0	-1.0	11.1	1.9	0.9	3.7
North America	0.1	0.4	41.5	3.2	1.0	96.3	1.4	0.7	10.2
Latin America and the Caribbean	3.9	-0.2	95.7	6.0	4.2	75.9	2.5	0.8	11.0
Developed countries	2.3	-0.1	33.2	3.0	-2.2	79.1	2.0	0.2	10.6
Developing countries	6.6	1.8	125.3	9.5	1.8	97.1	2.0	1.3	23.2
World	2.5	-0.1	58.0	4.6	0.1	89.6	2.0	1.0	17.9

[1] Includes arable and permanent cropland.
Source: FAO(2006b).

Table 2

Total calories, protein and fat intake and contribution of animal-derived foods in selected regions and countries

Region/Country	Total: Calories/capita/day (Number)	Total: Protein/capita/day (g)	Total: Fat/capita capita/day (g)	Percentage from animal products: Calories/capita/day	Percentage from animal products: Protein/capita/day	Percentage from animal products: Fat/capita/day
Commonwealth of Independent States	2 793	81	73	21	45	56
North Africa	3 203	88	65	8	21	28
North America	3 588	105	125	22	51	43
Sub-Sahara and South Africa	2 248	55	46	7	21	22
East and Southeast Asia	2 686	65	55	9	29	31
Eastern Europe	3 180	93	107	26	49	59
Latin America and the Caribbean	2 852	77	81	20	48	48
Near East	2 897	80	69	11	25	32
Oceania	2 971	94	115	29	63	54
South Asia	2 394	56	50	9	20	28
Western Europe	3 519	108	150	31	60	55
Australia	3 096	104	135	33	67	53
Brazil	3 006	81	92	22	52	50
China	2 942	82	86	20	37	58
India	2 423	56	52	8	19	25
Developed countries	**3 304**	**100**	**122**	**26**	**56**	**51**
Developing countries	**2 651**	**68**	**65**	**13**	**31**	**41**
World	**2 792**	**75**	**77**	**17**	**38**	**45**

Note: Three-year averages 2000–2002.
Source: FAO (2006b).

Table 3

Grassland area and share of total land covered by grassland in selected regions and countries

Region/Country	Total area of grassland	Percentage of total area as grassland
	(km²)	
North America	7 970 811	41.1
Latin America and the Caribbean	7 011 738	34.2
Western Europe	1 216 683	32.5
Eastern Europe	293 178	25.2
Commonwealth of Independent States	6 816 769	31.1
West Asia and North Africa	1 643 563	13.6
Sub-Saharan Africa and South Africa	7 731 638	31.5
South Asia	661 613	14.9
East and Southeast Asia	5 286 989	32.9
Oceania	5 187 147	58.1
Australia	4 906 962	63.6
China	3 504 907	37.3
India	371 556	11.7
Brazil	2 179 466	25.6
Developed Countries	**19 803 555**	**35.4**
Developing Countries	**18 369 118**	**24.0**
World	**38 172 673**	**28.8**

Source: Own calculation.

Table 4

Estimated net primary productivity in areas dominated by pasture

Region/Country	Mean net primary productivity	Area below 1200 (gr Carbon per m² and year)		Area above 1200 (gr Carbon per m² and year)	
		km²	*%*	*km²*	*%*
Commonwealth of Independent States	726.5	3 057 780	96.7	105 498	3.3
Latin America and the Caribbean	1254.6	2 297 740	47.4	2 548 350	52.6
Western Europe	948.8	766 276	72.4	291 848	27.6
West Asia and North Africa	637.0	1 800 730	92.7	142 480	7.3
Sub-Saharan Africa and South Africa	1226.1	5 066 060	42.8	6 777 050	57.2
South Asia	708.2	224 012	79.0	59 504	21.0
East and Southeast Asia	1158.1	652 412	43.0	863 624	57.0
North America	718.5	4 090 920	90.9	411 074	9.1
Eastern Europe	1080.4	152 280	72.0	59 261	28.0
Oceania	1189.3	143 905	58.3	102 736	41.7
Australia	1065.6	3 895 680	69.4	1 721 570	30.6
Brazil	1637.7	37 424	1.3	2 893 640	98.7
India	385.9	131 927	93.8	8 682	6.2
China	774.5	2 644 020	86.8	402 534	13.2
Developed	**871.0**	**12 473 500**	**79.8**	**3 153 290**	**20.2**
Developing	**1153.1**	**12 486 800**	**48.5**	**13 233 500**	**51.5**
World	**1046.5**	**24 960 300**	**60.4**	**16 386 790**	**39.6**

Note: Summary of Map 4, Annex 1.
Source: Own calculation.

Table 5
Current dominant land use in areas with high suitability for pasture but no current use as pasture

Region/Country	Forest	Forest	Cropland	Cropland	Urban	Urban
	km²	*%*	*km²*	*%*	*km²*	*%*
Commonwealth of Independent States	3 381 180	65.6	1 608 240	31.2	166 923	3.2
Latin America and the Caribbean	3 375 720	87.3	432 466	11.2	60 685	1.6
Western Europe	825 342	46.5	747 410	42.1	201 770	11.4
West Asia and North Africa	40 782	21.4	134 138	70.3	15 933	8.3
Sub-Saharan Africa and South Africa	3 642 730	87.9	442 489	10.7	58 440	1.4
South Asia	51 925	19.1	205 745	75.9	13 486	5.0
East and Southeast Asia	2 167 580	64.1	1 124 630	33.2	91 498	2.7
North America	2 515 240	51.4	2 172 750	44.4	203 408	4.2
Eastern Europe	334 619	36.5	517 651	56.5	64 671	7.1
Oceania	362 790	95.9	13 080	3.5	2 294	0.6
Australia	390 805	79.5	88 358	18.0	12 467	2.5
Brazil	4 766 500	95.3	126 222	2.5	107 969	2.2
India	186 840	22.9	595 042	72.9	34 553	4.2
China	873 628	42.4	1 047 920	50.9	138 976	6.7
Developed	**7 748 680**	**57.0**	**5 205 720**	**38.3**	**650 239**	**4.8**
Developing	**15 161 600**	**76.8**	**4 044 780**	**20.5**	**523 734**	**2.7**
World	**22 910 280**	**68.7**	**9 250 500**	**27.8**	**1 173 973**	**3.5**

Note: Summary of Map 12, Annex 1.
Source: Own calculation.

Table 6
Poultry population, density on agricultural land and ratio to human population in selected regions and countries

Region/Country	No. of animal	No. of animal per agricultural area	No. of animal per human population
	('000 head)	*(head/ha)*	*(head per head)*
North America	2 058 729	4.3	6.7
Latin America and the Caribbean	2 255 899	2.2	4.5
Western Europe	1 097 990	7.5	2.8
Eastern Europe	231 172	3.6	1.9
Commonwealth of Independent States	558 194	1.0	2.0
West Asia and North Africa	1 263 426	2.8	3.3
Sub-Saharan Africa	862 304	0.9	1.4
South Asia	700 772	1.7	0.5
East and Southeast Asia	5 994 579	4.4	3.1
Oceania	111 857	0.1	3.7
Australia	86 968	0.2	4.7
China	3 830 469	6.9	3.1
India	377 000	2.1	0.4
Brazil	877 884	3.3	5.3
Developed	**4 518 867**	**2.5**	**3.5**
Developing	**10 627 741**	**3.3**	**2.3**
World	**15 146 608**	**3.0**	**2.6**

Source: Own calculation.

Table 7
Pig population, density on agricultural land and ratio to human population in selected regions and countries

Region/Country	No. of animals	No. of animals per agricultural area	No. of animals per human population
	('000 head)	*(head/ha)*	*(head per head)*
North America	73 017	0.15	0.24
Latin America and the Caribbean	76 793	0.10	0.15
Western Europe	124 617	0.85	0.32
Eastern Europe	40 177	0.62	0.33
Commonwealth of Independent States	31 160	0.06	0.11
West Asia and North Africa	665	0.00	0.00
Sub-Saharan Africa	20 480	0.02	0.03
South Asia	14 890	0.07	0.01
East and Southeast Asia	528 673	0.66	0.27
Oceania	5 509	0.01	0.18
Australia	2 733	0.01	0.15
China	452 215	0.82	0.36
India	13 867	0.08	0.01
Brazil	32 060	0.12	0.19
Developed	**285 215**	**0.16**	**0.22**
Developing	**632 420**	**0.20**	**0.14**
World	**917 635**	**0.18**	**0.16**

Source: Own calculation.

Table 8
Cattle population, density on agricultural land and ratio to human population in selected regions and countries

Region/Country	No. of animals	No. of animals per agricultural area	No. of animals per human population
	('000 head)	*(head/ha)*	*(head per head)*
North America	110 924	0.23	0.36
Latin America and the Caribbean	357 712	0.46	0.71
Western Europe	84 466	0.58	0.21
Eastern Europe	16 042	0.25	0.13
Commonwealth of Independent States	58 395	0.10	0.21
West Asia and North Africa	31 759	0.07	0.08
Sub-Saharan Africa	213 269	0.21	0.35
South Asia	246 235	1.09	0.19
East and Southeast Asia	152 578	0.19	0.08
Oceania	37 796	0.08	1.26
Australia	27 726	0.06	1.49
China	103 908	0.19	0.08
India	191 218	1.06	0.20
Brazil	177 204	0.67	1.07
Developed	**326 830**	**0.18**	**0.25**
Developing	**983 781**	**0.31**	**0.22**
World	**1 310 611**	**0.26**	**0.22**

Source: Own calculation.

Table 9

Small ruminant population, density on agricultural land and ratio to human population in selected regions and countries

Region/Country	No. of animals	No. of animals per agricultural area	No. of animals per human population
	('000 heads)	*(head/ha)*	*(head per head)*
North America	9 132	0.02	0.03
Latin America and the Caribbean	115 514	0.15	0.23
Western Europe	121 574	0.83	0.31
Eastern Europe	20 902	0.32	0.17
Commonwealth of Independent States	59 649	0.11	0.21
West Asia and North Africa	227 378	0.50	0.59
Sub-Saharan Africa	370 078	0.37	0.60
South Asia	298 822	1.33	0.23
East and Southeast Asia	345 716	0.43	0.18
Oceania	153 302	0.32	5.11
Australia	112 202	0.25	6.03
China	289 129	0.52	0.23
India	181 300	1.00	0.19
Brazil	24 008	0.09	0.14
Developed	**400 136**	**0.22**	**0.31**
Developing	**1 322 038**	**0.42**	**0.29**
World	**1 722 175**	**0.34**	**0.29**

Source: Own calculation.

Table 10

Maize trade at regional level: 2001 to 2003 average and increment over the previous 15 years

From	Asia		Sub-Saharan Africa		North Africa		EU-15		Eastern Europe	
To	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		(%)		(%)		(%)		(%)
Asia	**11 669**	**853.1**	193.6	207.3	0.6	-	8.8	-92.0	293.3	82.9
Sub-Saharan Africa	220.5	574.3	**759.6**	**94.7**	0.1	-	26.5	-54.6	6.7	-14.1
North Africa	41.8	386.0	1.7	-	**43.4**	**-**	24.6	-92.1	83	-7.0
EU-15	6.8	-44.3	4.9	345.5	0.2	-	**8 837.4**	**41.7**	806.5	257.6
Rest of Western Europe	0	-100.0	0.1	-	0.8	-	20.4	-87.6	**38.6**	**-38.3**
Eastern Europe	0.5	-98.7	0.3	-	0	-	64.1	32.2	892.9	237.2
Baltic and CIS	6.7	-99.4	0.2	-	0	-	6	-88.0	130	-69.1
North America	0.3	-	0.2	-	0	-	0.7	-56.3	2.5	733.3
South America	0.2	-100.0	0.6	-90.3	0	-	0.3	-76.9	0	-
Central America and the Caribbean	16.7	53.2	1.7	-	0	-	0.1	-99.8	0	-100.0
Oceania	2.6	-99.8	0	-	0	-	0	-100.0	0	-

From	Baltic and CIS		North America		South America		Cntral America and the Caribbean		Oceania	
To	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		(%)		(%)		(%)		(%)
Asia	79.1	n.a.	24 120	13.0	6 631.8	362.8	0	-	23.6	-51.3
Sub-Saharan Africa	0.3	n.a.	404.9	180.4	525.8	879.1	7.3	-	3	-
North Africa	113.9	n.a.	5 791.7	143.9	2 347.4	452.3	0	-	0	-
EU-15	45.9	n.a.	68.6	-97.6	2 530.5	276.7	0	-	0.1	-50.0
Rest of Western Europe	0.5	n.a.	45.7	182.1	164.3	466.6	6.7	-	0	-
Eastern Europe	n.a.	10.7	-98.1	201	104.3	0	-	0	-	
Baltic and CIS	**261**	**n.a.**	43.8	-99.2	7.8	-99.0	0	-	0	-
North America	n.a.	3 799.9	**998.2**	**56.7**	18.6	37	469.2	0	-	
South America	14.8	n.a.	2 815.9	138.8	**2 745.9**	**431.1**	4.3	-	0.2	-
Central America and the Caribbean	10.2	n.a.	9 162.2	147.4	131	-75.0	**19.4**	**-**	0	-
Oceania	0	n.a.	22.2	404.5	0	-	0	-	**23.1**	**50.0**

Note: n.a. – data unavailable for the period 1986 to 1988.
- : negligible volume traded for the 2001 to 2003 average.

Source: FAO (2006b).

Table 11

Soybean trade at the regional level: 2001–2003 average and increment over the previous 15 years

From	USA		Brazil		Argentina	
	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		(%)		(%)
Total production	73 424.7	49.1	43 829.5	172.1	30 614.7	287.5
Total exports	29 128.8	44.2	17 178.7	655.5	7 412.6	266.6
Destination by region						
Asia	16 935.3	127.0	6 305.8	1 813.7	6 207.1	7 342.6
Sub-Saharan Africa	6.2	-71.9	0	-100.0	19.5	-
North Africa	336.3	294.7	111.9	-	193.8	-
EU-15	5 587.9	-38.5	9 852.7	498.6	745.4	-37.4
Rest of Western Europe	19.1	-90.2	404	859.6	0.3	-99.1
Eastern Europe	45.4	-91.2	106.8	87.0	5.4	-93.1
Baltic states and CIS	65.6	-92.0	17.7	5 800.0	0	-100.0
North America	640.7	311.2	2.2	-	12.7	-
South America	213.5	-62.8	248.8	82 833.3	198.7	-
Central America and the Caribbean	4 563.4	279.1	128.7	4 190.0	29.8	33.6
Oceania	18.6	-41.9	0	-100.0	0	-

From	Paraguay		Canada		India		China	
	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		(%)		(%)		(%)
Total production	3 671.9	212.3	2 079.7	84.5	5 773.6	419.1	15 768.3	33.5
Total exports	2 019.1	103.1	671.8	233.9	83.3	-	263.9	-82.6
Destination by region								
Asia	14.3	-	344.7	353.0	83.1	3 362.5	253.9	-52.7
Sub-Saharan Africa	0.1	-	0.3	200.0	0	-	0	-
North Africa	0	-100.0	5.6	51.4	0	-	0	-
EU-15	62.5	-75.5	200.7	208.3	0	-	7.8	13.0
Rest of Western Europe	208.6	104.5	0	-100.0	0	-	0	-
Eastern Europe	0	-	1.1	-	0	-	0.1	-99.3
Baltic states and CIS	1.7	-	0.1	-99.5	0	-	0.3	-99.9
North America	0	-100.0	112.5	224.2	0.1	-	0.9	-
South America	1 383.8	1 176.6	0	-	0	-	0.6	-92.7
Central America and the Caribbean	348.1	234.7	6.3	-	0	-	0	-
Oceania	0	-	0.4	-	0	-	0.4	-

Note: n.a. : data unavailable for the period 1986 to 1988.
–: negligible volume traded for the 2001 to 2003 average.
Source: FAO (2006b).

Table 12

Soymeal trade at regional level: 2001–2003 average and increment over the previous 15 years

From	Asia		Sub-Saharan Africa		North Africa		EU-15		Eastern Europe	
To	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		*(%)*		*(%)*		*(%)*		*(%)*
Asia	**2 890.3**	**177.1**	0.1	-	0	-	30.7	-72.3	0	-
Sub-Saharan Africa	10.5	218.2	**6.8**	**-50.0**	8.8	-	13.5	-2.9	0	-
North Africa	41.3	3.8	0.2	-	**0**	**-**	27.5	-69.3	0	-
EU-15	7.7	-96.8	0.2	-	0	-	**4 417.9**	**38.2**	1.5	-
Rest of Western Europe	0.1	-99.7	0	-	0	-	143.1	530.4	**0**	**-**
Eastern Europe	1.5	-99.6	0	-	0	-	1 617.6	1 202.4	40.3	-
Baltic and CIS	3.7	-93.5	0	-	0	-	217.4	-14.6	3.4	-
North America	0.2	-96.8	0	-	0	-	0.7	250.0	0	-
South America	0.5	-	0.1	-	0	-	0.4	-50.0	0	-
Central America and the Caribbean	0	-100.0	0	-	0	-	0.3	-91.4	0	-
Oceania	3.7	208.3	0	-	0	-	27.4	6750.0	0	-

From	Baltic and CIS		North America		South America		Cntral America and the Caribbean		Oceania	
To	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		*(%)*		*(%)*		*(%)*		*(%)*
Asia	0	n.a.	2 122.9	196.9	6 361.9	-	0	-	0	-
Sub-Saharan Africa	0	n.a.	4.8	-94.5	532.8	366.1	0	-	0	-
North Africa	0	n.a.	421	10.0	1 298.7	714.2	0	-	0	-
EU-15	0	n.a.	345.7	-85.0	18 875.8	223.1	10.9	-	0	-
Rest of Western Europe	0	n.a.	2.2	450.0	36.1	163.5	0	-	0	-
Eastern Europe	0	n.a.	13.4	-93.2	851.9	-49.3	0	-	0	-
Baltic and CIS	**14**	**n.a.**	106	-77.2	9.8	-99.3	0	-	0	-
North America	1	n.a.	**764.4**	**3.5**	46.1	-	1.1	-57.7	0	-
South America	2	n.a.	324	-54.9	**1 912.8**	**-**	14.8	-	0	-
Central America and the Caribbean	0	n.a.	1 509.8	256.4	82.6	-54.6	**30.2**	**174.5**	0	-
Oceania	0	n.a.	322.8	701.0	190.3	-	0	-	**0.2**	**-75.0**

Note: n.a. : data unavailable for the period 1986 to 1988.
–: negligible volume traded for the 2001 to 2003 average.
Source: FAO (2006b).

Table 13

Bovine meat trade at regional level: 2001 to 2003 average and increment over the previous 15 years

From	Asia		Sub-Saharan Africa		North Africa		EU-15		Eastern Europe	
To	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		*(%)*		*(%)*		*(%)*		*(%)*
Asia	**271.1**	**330.3**	1.9	533.3	0.4	-	132.5	-45.8	0.3	-97.3
Sub-Saharan Africa	42.3	-	**48.7**	**-**	0.0	-	42.0	-79.5	0.0	-100.0
North Africa	9.7	-	1.3	-	**0.0**	**-**	2.4	-98.3	0.0	-100.0
EU-15	8.8	-	14.5	29.5	0.1	-	**1 514.4**	**6.7**	23.3	-57.2
Rest of Western Europe	0.9	-	2.0	-	0.0	-	9.4	-30.4	**0.2**	**-89.5**
Eastern Europe	0.6	-	0.0	-	0.0	-	24.3	-68.6	40.0	273.8
Baltic and CIS	31.7	-11.9	0.0	-	0.0	-	351.5	343.3	23.0	-63.5
North America	2.5	-	0.0	-	0.0	-	1.7	-98.1	0.1	-99.3
South America	0.2	-	0.0	-	0.0	-	0.6	-99.5	0.0	-100.0
Central America and the Caribbean	0.1	-90.0	0.0	-	0.0	-	1.2	-90.3	0.0	-100.0
Oceania	0.4	-	0.0	-	0.0	-	0.2	-98.2	0.1	0.0

From	Baltic and CIS		North America		South America		Cntral America and the Caribbean		Oceania	
To	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		*(%)*		*(%)*		*(%)*		*(%)*
Asia	0.2	n.a.	680.5	260.6	270.1	108.6	1.0	-60.0	686.5	173.3
Sub-Saharan Africa	0.0	n.a.	0.3	0.0	21.9	-28.9	0.0	-	3.6	-
North Africa	0.0	n.a.	8.2	-	132.9	-	0.0	-	4.5	-
EU-15	0.8	n.a.	3.5	-65.0	390.5	84.1	0.0	-	11.1	-31.9
Rest of Western Europe	0.0	n.a.	1.4	75.0	9.0	-13.5	0.0	-	2.5	177.8
Eastern Europe	0.0	n.a.	0.4	-	52.3	-	0.0	-	2.2	-
Baltic and CIS	**236.3**	**n.a.**	5.4	-	53.1	-	0.0	-	6.9	-
North America	0.0	n.a.	**520.8**	**416.7**	161.3	86.5	42.5	-14.8	903.7	14.3
South America	0.0	n.a.	3.4	-87.7	**208.9**	**139.3**	2.0	-	0.1	-
Central America and the Caribbean	0.0	n.a.	333.8	2 110.6	16.3	3.2	**29.1**	**627.5**	19.8	219.4
Oceania	0.0	n.a.	1.4	75.0	0.6	500.0	0.0	-	**40.6**	**50.4**

Note: n.a. : data unavailable for the period 1986 to 1988.
–: negligible volume traded for the 2001 to 2003 average.
Source: FAO (2006b).

Table 14

Poutry meat trade at regional level: 2001 to 2003 average and increment over the previous 15 years

From	Asia		Sub-Saharan Africa		North Africa		EU-15		Eastern Europe	
To	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		*(%)*		*(%)*		*(%)*		*(%)*
Asia	**915.9**	**526.5**	1.6	-	0.6	500.0	291	48.4	6.7	-53.5
Sub-Saharan Africa	7.6	-	**10.9**	**-**	0.1	-	215.9	149.3	0.4	-60.0
North Africa	0.2	100.0	0	-	**0**	**-**	2.9	-62.8	0.2	-92.3
EU-15	194	-	2.7	-	0.6	-	**1 836.9**	**265.6**	143.2	130.6
Rest of Western Europe	9	718.2	0	-	0	-	25.5	9.9	**8.6**	**-18.9**
Eastern Europe	19.9	-	0	-	0	-	123.3	-	47.8	414.0
Baltic and CIS	28.8	-	0	-	0	-	304.5	-	26.8	-82.5
Norh America	2.9	314.3	0.1	-	0	-	1.7	54.5	0	-100.0
South America	2.6	-	0.2	-	0	-	0.8	-87.5	0	-100.0
Central America and the Caribbean	1.1	-	0	-	0	-	20	-6.5	0	-100.0
Oceania	0.8	-38.5	0	-	0	-	4.8	6.7	0.2	100.0

From	Baltic and CIS		North America		South America		Cntral America and the Caribbean		Oceania	
To	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase	'000 tonnes	Increase
		(%)		*(%)*		*(%)*		*(%)*		*(%)*
Asia	0.1	n.a.	946	382.9	927	378.1	1.1	-	9.7	781.8
Sub-Saharan Africa	0	n.a.	104.9	-	115.9	-	0	-	9.9	-
North Africa	0	n.a.	1.9	-90.9	2.8	27.3	0	-		-
EU-15	0.1	n.a.	48.9	304.1	375.4	-	0	-	0	-
Rest of Western Europe	0	n.a.	2.6	420.0	3	-57.7	0	-	0	-
Eastern Europe	0.3	n.a.	122.2	-	30.5	-	0	-	0	-
Baltic and CIS	**34.2**	**n.a.**	1022.5	-	225.2	-	0	-	0.2	-
Norh America	0.1	n.a.	**164.1**	**374.3**	2.5	-	3.8	-	0	-
South America	0	n.a.	43.5	-	**31.6**	**212.9**	0.4	-	0	-
Central America and the Caribbean	0	n.a.	502.6	570.1	43.5	559.1	**5.2**	**-**		-
Oceania	0	n.a.	25.2	334.5	1.5	-	0	-	**5.7**	**159.1**

Note: n.a. : data unavailable for the period 1986 to 1988.
–: negligible volume traded for the 2001 to 2003 average.
Source: FAO (2006b).

Table 15

Major meat trade flows in the 2001-2003 period, their volume and related sea transport CO_2 emission

Country of origin	Destination countries	Traded quantity (10^3 tonnes)	Fossil fuel CO_2 emission (10^3 tonnes)
Bovine meat			
United States	Canada, Japan, Hong Kong, Korea, Mexico	1 000	34
Australia	USA, Canada, Japan, South Korea	1 055	61
Brazil	Hong Kong, EU, Saudi Arabia, USA, Egypt	390	28
Canada	USA, Mexico	497	7
New Zealand	USA, Canada	418	20
	share of global trade:	**60%**	**150**
Poultry meat			
USA	China, Hong Kong, Japan, South Korea, Russia, Mexico, Canada	2 093	137
Brazil	Japan, Hong Kong, Russia, Saudi Arabia, EU	921	82
European Union	Russia, Saudi Arabia	342	9
China	Japan	364	4
Thailand	EU, Japan	381	20
Hong Kong	China	660	5
	share of global trade:	**63%**	**257**
Pig meat			
Canada	Japan, USA	543	14
European Union	Japan, Russia	473	34
United States	Japan, Mexico	400	12
Brazil	Hong Kong, Russia	247	23
China	Hong Kong, Russia	133	1
	share of global trade:	**53%**	**85**

Source: Data on meat trade flows – FAO (2006b).

Table 16

Possible contribution of livestock to the extinction of species through habitat loss or degradation

Species	Description
ANIMALS	**Amphibia**
Atelopus longirostris	Endemic to the northwestern versant of the Andes of Ecuador, in the provinces of Esmeraldas, Imbabura, Cotopaxi, and Pichincha at 500–2 500 m above sea level (asl). It was a terrestrial species, living in lowland and montane tropical rain forests. The decline of the population is unexplained, and is possibly due to chytridiomycosis. Other possible factors include climate change, pollution and habitat loss, though these are unlikely to explain the level of decline that has been observed.
Atelopus vogli	Endemic to the Pozo del Diablo in the river Güey, on the southern versant of the Cordillera de la Costa, Venezuela. The original habitat (humid forest) at the type locality has been drastically modified by repeated clearing and burning. A savanna-like environment remains. It is presumed that the area formerly supported a semi-deciduous forest. The species is believed to have become extinct following drastic modification of habitat for agricultural use.
Eleutherodactylus chrysozetetes	This species was endemic to Quebrada de Oro in the Rio Viejo, Honduras, at 880–1 130 m asl. I It was found along streams in premontane wet forest. It was probably unable to survive the severe degradation of its habitat that has taken place. The threats include deforestation as a result of agricultural and livestock encroachment, human settlements, logging, fires and landslides. Chytridiomycosis may have also contributed.
Eleutherodactylus milesi	Endemic to the premontane wet forest and lower montane wet forest of mountains of west and northwest of Honduras at 1 050–1 720 m asl. It was clearly adversely affected by habitat description caused by subsistence agriculture and by chytridiomycosis.
Rheobatrachus silus	An Australian endemic species, lived in rainforest, wet sclerophyll forest and riverine gallery open forest and restricted to elevations between 350–800 m asl in the Blackall and Conondale Ranges in southeast Queensland. The reason(s) for the disappearance of this species remains unknown. Its habitat is currently threatened by feral pigs, invasion of weeds (especially mistflower Ageratina riparia), and altered flow and water quality due to upstream disturbances and possible to chytridiomycosis.
Rheobatrachus vitellinus	An Australian endemic was found exclusively in undisturbed rainforest in Eungella National Park, mid-eastern Queensland at altitudes of 400–1 000 m asl (Covacevich and McDonald 1993). The extent of occurrence of the species was less than 500 km². The cause(s) of the population decline remain unknown. Possible reasons habitat destruction by seasonal fire, fragmentation, weeds, surface water extraction and chytridiomycosis.
	Birds
Cabalus modestus	Endemic to Chatham, Mangere and Pitt Islands, New Zealand. Its extinction was presumably caused by predation by rats and cats, habitat destruction to provide sheep pasture (which destroyed all of the island's bush and tussock grass by 1900), and from grazing by goats and rabbits.
Caracara lutosa	This species was endemic to Isla Guadalupe, Mexico. The island was once heavily vegetated, but grazing by goats has almost entirely denuded it. The primary cause of the species' decline was direct decimation by settlers.
Coturnix novaezelandiae	Endemic to open habitats, especially grass-covered downs, on North, South and Great Barrier Islands, New Zealand. It was considered fairly common until the mid-19th century, but declined rapidly to extinction by 1875. Extinction was caused by large-scale burning, predation by dogs, cats and rats, and grazing by sheep and by diseases spread by introduced gamebirds.

(cont.)

Table 16 *(cont.)*

Possible contribution of livestock to the extinction of species through habitat loss or degradation

Species	Description
Drepanis funerea	Endemic to the forest understorey on Molokai, Hawaii, USA. Its extinction was probably largely caused by the destruction of its understorey habitat by introduced cattle and deer, and predation by rats and mongooses.
Moho bishopi	Endemic to forest in the Hawaiian Islands, USA. Habitat destruction caused by conversion to agriculture and grazing by feral mammals inevitably initiated the species' decline, with introduced black rat Rattus rattus and the spread of disease carried by introduced mosquitoes blamed for the population decline.
Myadestes myadestinus	Endemic to Kaua'i in the Hawaiian Islands (USA), where it was probably restricted to dense montane forest. It was the most common of the forest birds. Disease carried by introduced mosquitoes and the destruction and degradation of forests are likely to have been the chief causes of extinction. The advance of feral pigs into pristine upland forests degraded habitat and facilitated the spread of mosquitoes. Competition with introduced birds may have exacerbated the problems faced by this species.
Sceloglaux albifacies	Endemic to New Zealand with the nominate race on the South and Stewart Islands. The species roosted and nested among rocks in open country and on forest edge. Causes of the species' extinction are obscure, possibly habitat modification through grazing or burning, or predation by introduced rats.
Psephotus pulcherrimus	Found in open savanna woodland and shrubby grassland in central and southern (and possibly northern) Queensland and northern New South Wales, Australia. Its decline was probably caused by a reduction of its food supply (native grass seeds) due to drought, overgrazing, altered fire frequencies and the spread of prickly pears, with disease, trapping and egg-collecting, predation of nests by introduced and native species and clearance of eucalypti by ring barking also contributing.
PLANTS	**Magnoliopsida**
Nesiota elliptica	Endemic to St Helena island. A small tree that was known to be pollinated by an endemic syrphid fly, which also visits other endemic trees. The threat to this species was loss of habitat through felling for timber and to make way for plantations. Humans have exploited the island's resources for over 450 years, destroying much of the native vegetation through deforestation for timber and agriculture, and the grazing of introduced goats.
Cyanea marksii	A small palm-like tree recorded in the rainforest to Kona District on Hawaii. The forests and rare plants of South Kona are threatened by cattle grazing, logging, feral pigs, and alien plants. The plants are also naturally threatened by lava flows from Mauna Loa.
Melicope haleakalae	A small tree or shrub last seen in 1919 at Ukuele, on Maui. Habitat and Ecology Was found in rainforest at 1,220 m. Known only from the northwestern flank of Haleakala, Maui. The status of this species is unclear; it may be more common than currently thought. Threats include feral pigs and alien plants.
Melicope paniculata	Endemic to Lihue Ditch Trail at 875 m and from Waihiawa Bog at 580 m. Threatened by feral pigs, goats, alien plants.
Oldenlandia adscenionis	Found on Green Mountain on the northern and western slopes between 356 m and 680 m. This species was very susceptible to grazing mammals. Introduced plants have completely replaced the original vegetation communities and livestock (sheep and donkeys now present) goats were likely to have been historically responsible for the decline.

(cont.)

Table 16 *(cont.)*

Possible contribution of livestock to the extinction of species through habitat loss or degradation

Species	Description
Wikstroemia skottsbergiana	It appears the species may be extinct from its only locations in Hanalei and Kauhao Valleys on Kauai.The rare native plants of Hanalei Valley and the Wahiawa Mountains are threatened by feral pigs and alien plants.
Wikstroemia villosa	Was known from the windward side of Hakeakala on East Maui and from two collections from the ridges in Wailuku Valley on West Maui. A montane rainforest species. Parts of its range have been converted into pasture. Major threats are feral pigs and alien plants. It was also possibly threatened by deer, cattle, and feral goats.
	Liliopsida
Sporobolus durus	The introduction of species such as Melinis minutiflora (pioneer grass widely used for grazing) are likely to be responsible for the decline. M minutiflora as an easily established (by sowing) and productive grass of acceptable nutritive value; also used for soil conservation on steep slopes with poor soils. Resistant to drought but not to fire or water logging. Continues to grow throughout the year with some rainfall. Must be well established before grazing. Palatable to cattle once they become used to the smell.

Source: Compiled from IUCN, NatureServe, BirdLife International and ARKive.

Annex 3

Methodology of quantification and analysis

Annex 3

Methodology of quantification and analysis

3.1 Trends in land use for livestock

Methodology developed to assess arable land use for livestock

Derived from *nations' water footprints*, by Chapagain and Hoekstra, 2004.

Categories of arable crops included in the analysis are:

- cereals, e.g. wheat, maize, barley, buckwheat, sorghum, rye, millet, oats, mixed grains, rice paddy;
- oilseeds and fruits for oil, e.g. soybeans, sunflower, safflower, rapeseed, linseed, groundnuts, cottonseed, mustard seed, hempseed, coconut, oil palm fruit, olives, kapok fruit;
- roots and vegetables, e.g. cassava, yams, potatoes, sweet potatoes, cabbages, pumpkins, sugarcane, lupins, vetches, carobs, plantains;
- pulses, e.g. peas, beans, lentils ; and
- fruits, e.g. watermelons, apples, bananas, dates, citrus fruit.

The calculation differentiates crops fed directly (in their primary form) to livestock from those that are first processed, and for which by-products only are fed to livestock. Crop residues were not included because of data unavailability.

a) Direct feed products include primary crops obtained directly from the land and which do not undergo any real processing. Arable land area is obtained as the ratio of the feed element to the sum of the total supply utilization elements times the total area harvested .

b) By-products/derived feed products include:

- cake from processing of oilseeds and fruit for oil;
- bran, flour (maize and wheat), gluten (maize and wheat) and germ (maize and wheat) from processing of cereals;
- citrus pulp; and
- molasses from the processing of sugarcane and sugar beet

The quantity of harvested crop that is processed is first obtained from statistical databases. The arable land related to the amount of processed harvest is then calculated using the same technique as described above for direct feed.

As a next step we need to find out what fraction of this land can be ascribed to the production of by-products for feed. To obtain this we further multiply the arable land area related to processed product by the value fraction of the by-product with comparison to all product(s) from processing. The result is the amount of land attributed to the by-product.

The following data sources were used:

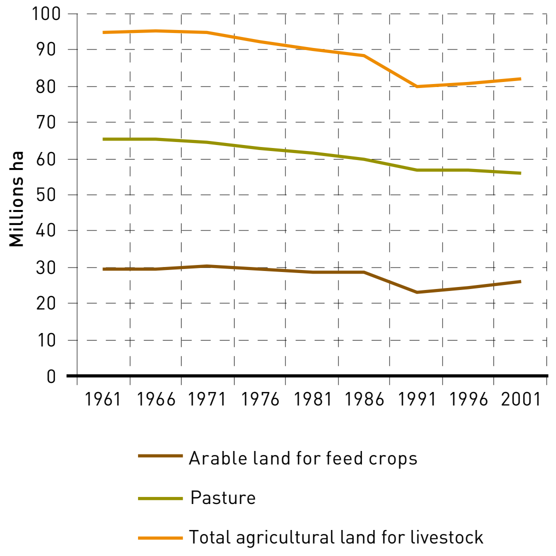
- FAO Supply utilization accounts (SUAs) the accounts give a detailed breakdown of the amount of crop supplied and how much is utilized in the different uses such as food, feed, waste, processing, seed and other, in a given period. The accounts also specify the area harvested, yield, production and area sown (FAO, 2006b). International commodity prices for primary products and derived products: (Chapagain and Hoekstra, 2004 and FAO international commodity prices)
- Commodity/product trees: These give the extraction rates/product fractions i.e. the

amount (in percentage terms) of the processed product concerned obtained from the processing of the parent product (FAO commodity Trees and Chapagain and Hoekstra, 2004).

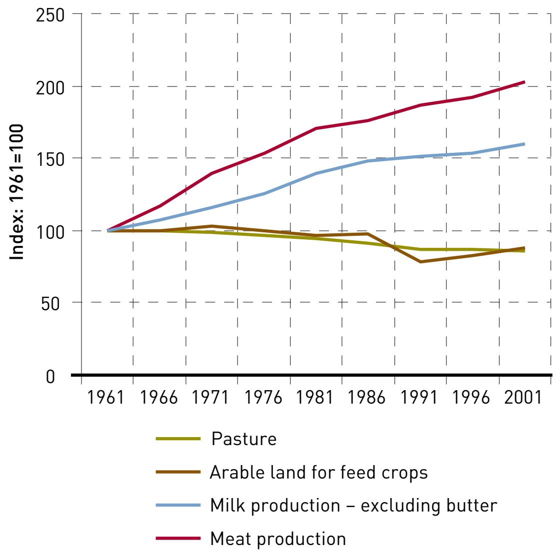
Selected results

EU –15

EU–15: trends in land-use area for livestock production

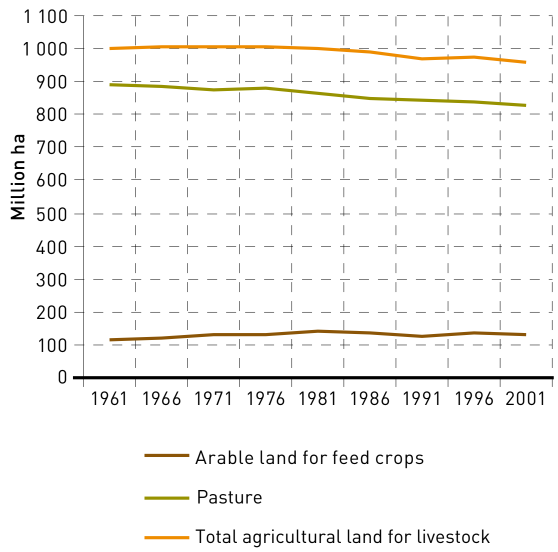


Trends in land-use area for livestock production and production of meat and milk

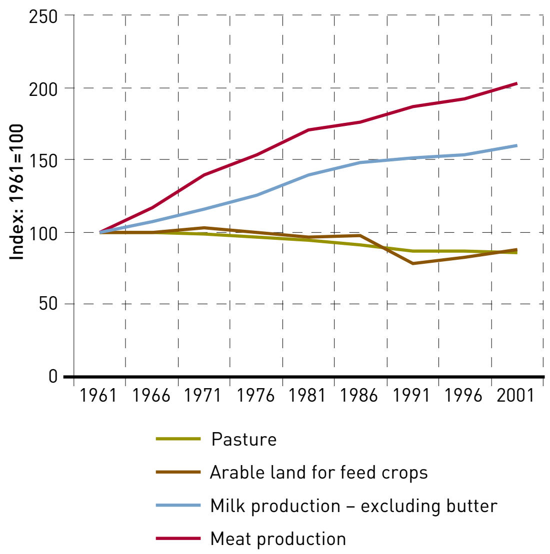


OECD

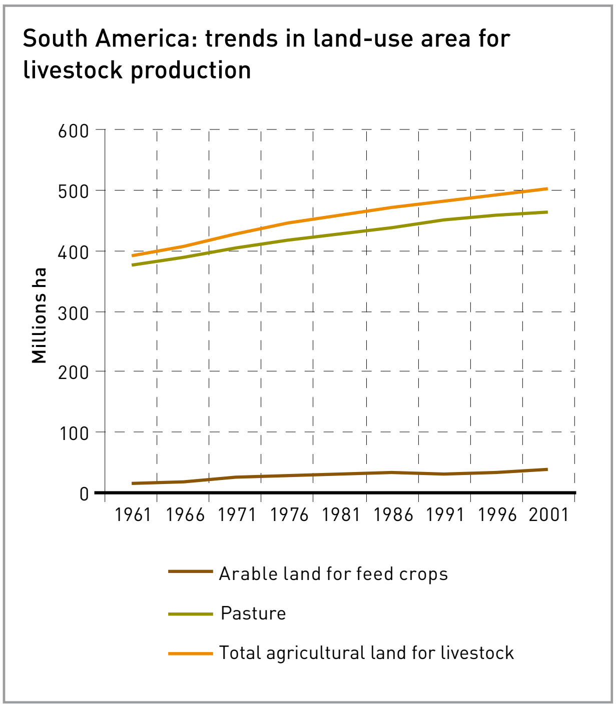
OECD: trends in land-use area for livestock production

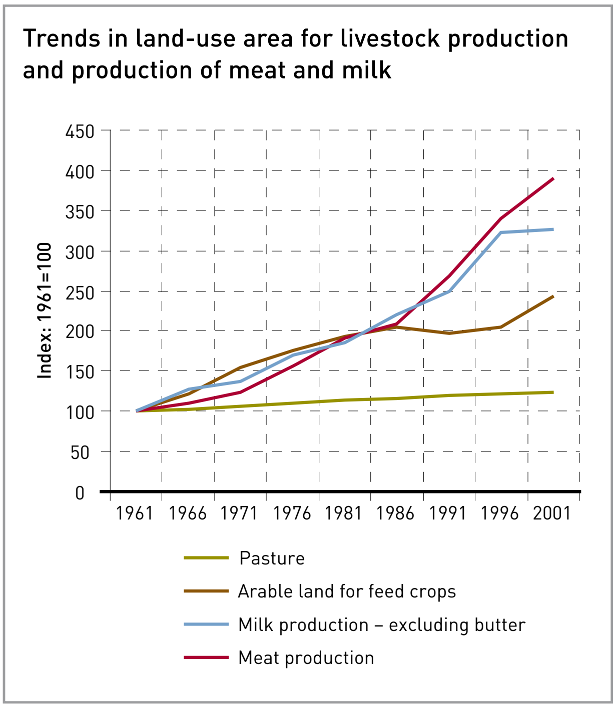


Trends in land-use area for livestock production and production of meat and milk

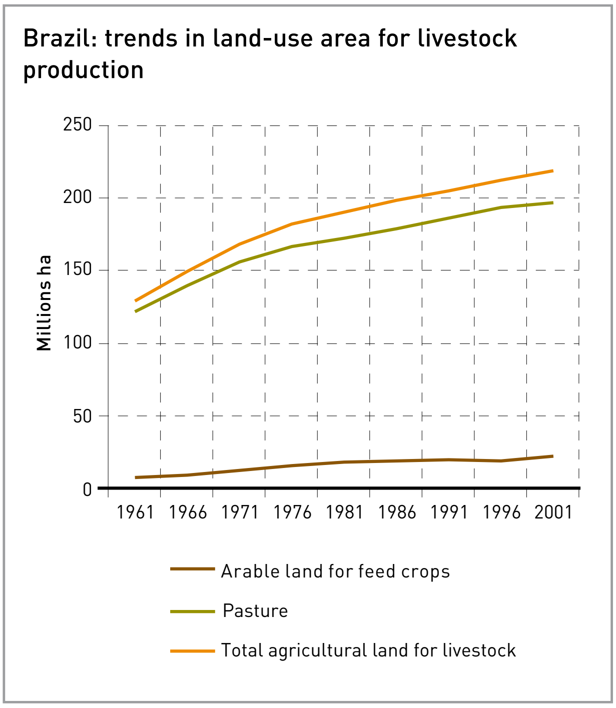


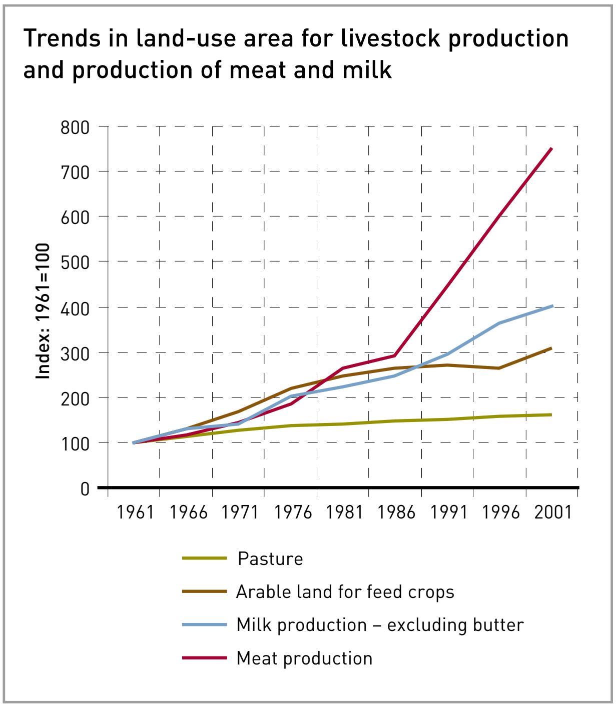
SOUTH AMERICA





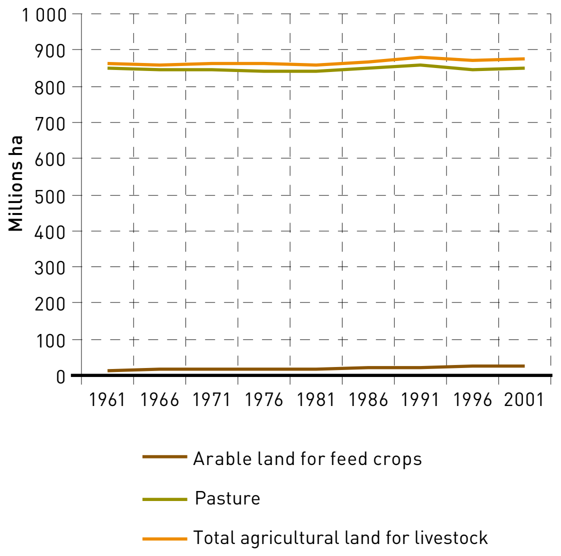
BRAZIL



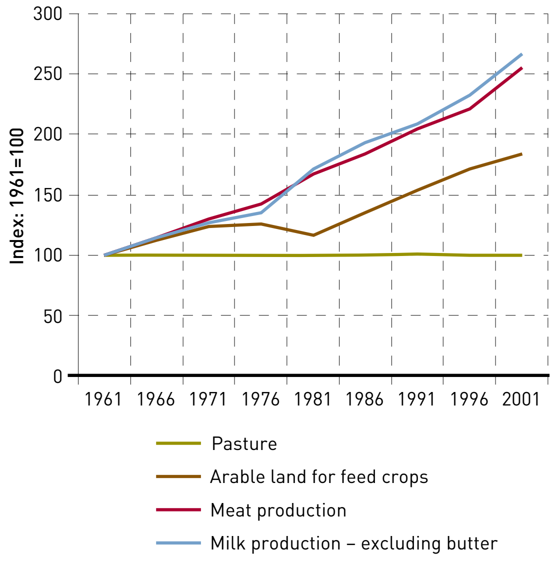


SUB-SAHARAN AFRICA

Sub-Saharan Africa: trends in land-use area for livestock production

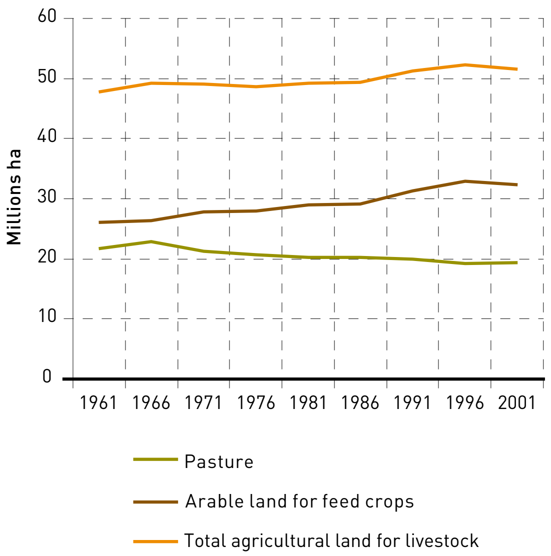


Trends in land-use area for livestock production and production of meat and milk

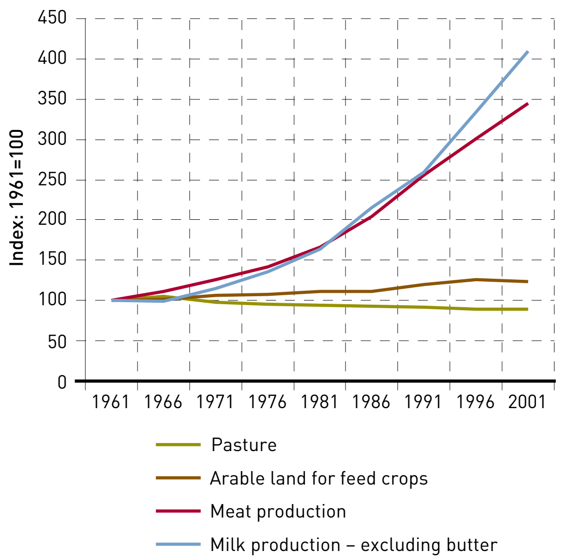


SOUTH ASIA

South Asia: trends in land-use area for livestock production

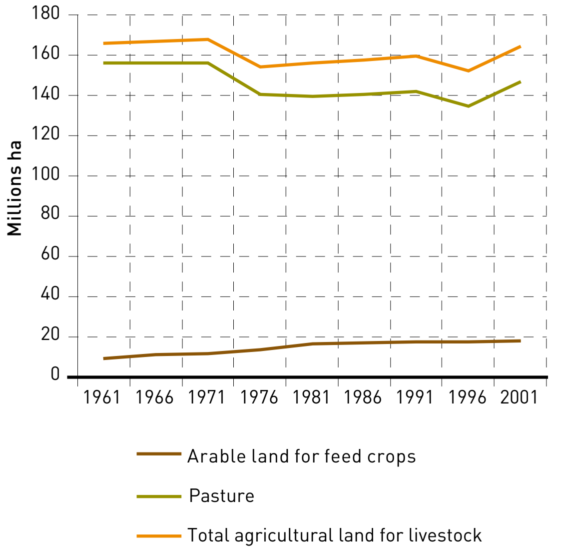


Trends in land-use area for livestock production and production of meat and milk

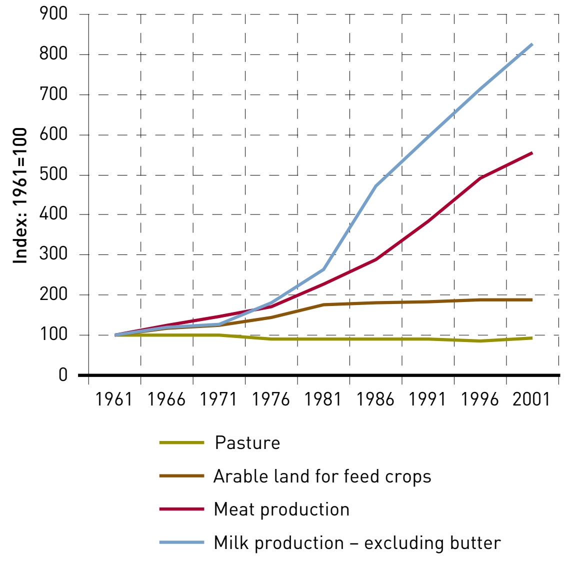


EAST AND SOUTHEAST ASIA

East and Southeast Asia: trends in land-use area for livestock production

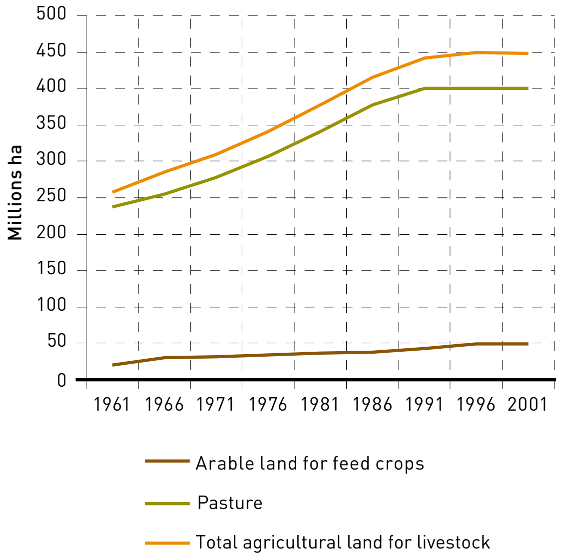


Trends in land-use area for livestock production and production of meat and milk

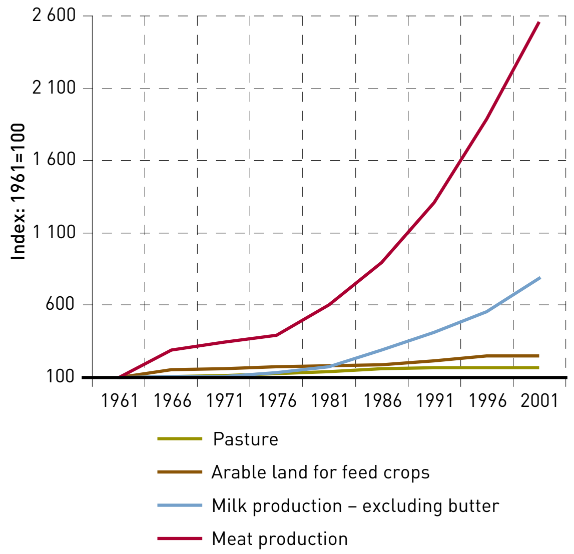


CHINA

China: trends in land-use area for livestock production

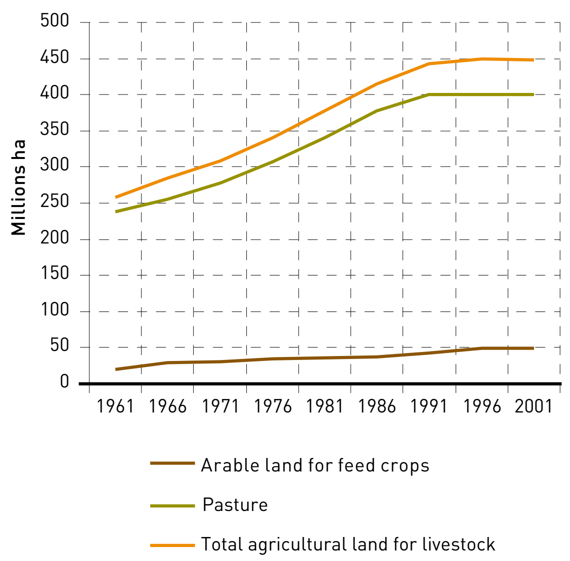


Trends in land-use area for livestock production and production of meat and milk

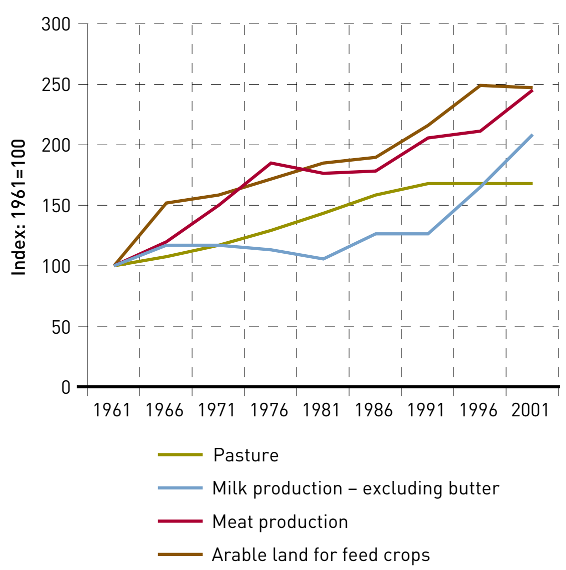


OCEANIA

Oceania: trends in land-use area for livestock production



Trends in land-use area for livestock production and production of meat and milk



3.2 Current enteric fermentation methane emissions per production system, species and region

Much of the information used by the IPCC to establish region-specific default emission factors for methane was published twenty years ago. As described in Chapter 2, livestock production characteristics in many regions have evolved considerably since then. An assessment was made for this report to evaluate the resulting discrepancy. The IPCC Tier 2 methodology was used to derive enteric fermentation emission factors for the most important animal categories dairy and other cattle (Houghton *et al.*, 1997). The following data were required to derive the average daily energy intake of the animal, which is then combined with a methane conversion factor for specific feed types:

- live weight;
- average daily weight gain (not relevant for dairy cattle);
- feeding situation (confined, grazing good pasture, extensive grazing);
- milk production per day;
- work performed per day (draft animals not relevant for dairy cattle);
- proportion of cows that give birth per year; and
- feed digestibility.

For each region and production system, average milk yield per dairy cow and mean livestock weight for other cattle were taken from the FAO database. Other data required were derived from the IPCC Guidelines Reference Manual (Houghton *et al.*, 1997), Table A3.1, appropriate to each world region. Feed digestibility and methane conversion rates were derived both from Houghton *et al.* (1997) and the EPA Livestock Analysis Model.

For all other livestock types, the Tier 1 approach was used as more detailed activity data were not available and the sources are relatively minor compared with cattle.

Therefore, for buffaloes, sheep, goats and pigs, default emission factors as given in Table 4-3 of the IPCC manual were used, with that for 'Developed countries' being used for 'Industrial

TABLE A3.1

Enteric fermentation emission factors (EF) for cattle (kilogram CH_4 per head per year) by production system and world region. Tier 2 based estimates compared to tier 1 emission factors

Region	Dairy cattle				Other cattle				
	Grazing	Mixed	Weighted EF	Tier 1 EF	Grazing	Mixed	Industrial	Weighted EF	Tier 1 EF
Sub-Saharan Africa	79	39	60	36	44	27	-	36	32
Asia excluding China and India	79	53	54	56	66	38	-	38	44
India	70	45	45	46	41	17	-	18	25
China	102	63	84	56	85	38	-	49	44
Central and South America	93	62	78	57	58	33	23	47	49
West Asia and North Africa	91	60	61	36	49	31	-	32	32
North America	115	100	100	118	50	33	26	35	47
OECD excluding North America	102	97	98	100	45	27	26	32	48
Eastern Europe and the CIS	-	59	59	81	-	45	24	41	56
Other developed	96	129	99	36	45	27	28	45	32

Source: Own calculations.

systems' where appropriate (e.g. for intensively reared swine in developing countries).

Table A3.1 allows us to compare the results with the currently used IPCC Tier 1 emission factors. In comparison, the main effects of using IPCC Tier 2 methodology to derive enteric fermentation methane emission factors for cattle have been:

- an increase in the weighted average emission factor for dairy cattle in most developing regions (by the proportion of animals associated with each livestock system); and
- a decrease for other cattle in OECD and transition regions.

The main reasons for these differences is better differentiation of both the feed digestibility and methane conversion factors associated with different feed types according to production system. For dairy cattle, IPCC default Tier 1 assumes feed digestibility of 60 percent for all regions except North America (65 percent) and India (55 percent), and a methane conversion factor of 6 percent for all regions.

For the Tier 2 approach, feed digestibility and methane conversion factors were estimated for the different production systems and world regions according to recommendations given by the US-EPA (EPA Ruminant Livestock). Following these, common feed digestibility for cattle ranges from 50 to 60 percent for crop by-products and rangelands; 60 percent to 70 percent for good pastures, good preserved forages, and grain supplemented diets; and 75 to 85 percent for high quality feedlot grain diets. The methane conversion factor for 'good quality feeds' is given as 6 percent, while that for 'poor quality feed', which might be taken to describe pastoral systems in most developing countries, is given as 7 percent. Therefore, the association of low feed digestibility and high methane conversion factor in pastoral systems, in developing countries, where both of these apply, has led to a greater emission factor being derived using Tier 2 methodology in these systems than those given under Tier 1. In addition, there were some differences in the default milk yields used to obtain the Tier 1 values and those derived from recent FAO statistics and used in the Tier 2 calculations. Obviously, great improvements to the estimates of emission factors could be made if more data on nutrition and production were available.

3.3 Current manure methane emissions per production system, species and region

As in the case of the enteric fermentation emission factors, the IPCC default manure methane emission factors were established some time ago and may not represent the current situation correctly. The structural changes in the livestock sector may have an important impact on overall methane emissions from manure.

Again an assessment was made for this report to evaluate the discrepancy: an IPCC Tier 2 approach was used to derive the methane from manure management emission factors for dairy cattle, other cattle and pigs (Houghton *et al.*, 1997). The emission factor per head was derived from the calculated volatile solids (VS) content of the manure per livestock type, together with an estimate of the methane-producing potential of the manure (Bo value) and a methane conversion factor, dependent on the manure management system.

Calculating VS required data for feed energy intake, digestibility and ash content of manure. For dairy cattle, the feed energy intake as calculated in the enteric fermentation emission factors was used, together with default IPCC values for digestibility and ash. For other cattle and pigs, default IPCC values for these parameters were used. For industrial pig systems in developing regions, we used values otherwise applied to developed countries. Emission factors were derived with the following assumptions for manure management system allocations:

- For cattle (dairy and other) in a grazing production system it was assumed that all manure management would be regarded as pasture/range management (i.e. 100 percent in that category).
- For 'other cattle' on an 'industrial' system it was assumed that all manure management would be regarded as 'drylot' (i.e. 100 percent in that category).
- The remaining cattle manure management categories (see Houghton *et al.*, 1997) were assumed to be associated with 'mixed' production systems, with the assumption that pasture/range was 15 percent of manure for mixed dairy systems and 20 percent for mixed beef systems.
- For pigs, responses to survey questionnaires were used together with the assumption that in developed countries, industrial systems would be predominantly slurry/lagoons with over one month of storage.
- For other livestock, default values (Houghton *et al.*, 1997) were used for appropriate systems ('developed' = 'industrial') and temperature regions. Again, this Tier 1 approach was used because less activity data were available for these livestock categories and they represented minor emission sources.

For the manure management methane emission factors, the IPCC Tier 2 methodology has again given estimates that are often greater than the Tier 1 defaults (Table A3.2), giving particularly large values for industrial systems. This is largely owing to the use of revised methane conversion factors for slurry storage systems as given by IPCC, 2000. These were increased from 10, 35 and 65 percent for cool, temperate and warm climates, respectively (being the values on which the Tier 1 default values are based) to 39, 45 and 72 percent for cool, temperate and warm climates, respectively. In addition, the feed digestibility characteristics, as described above, influenced the calculation of the manure volatile solids output per animal, on which the manure management methane emission factor is based.

The impact of the difference of course depends on the relative importance of the corresponding livestock populations, as well as whether Tier 1 factors are currently used (non-Annex 1, i.e. developing, countries). In this respect the increase of the estimated emission factor with respect to Tier 1 for cattle in Africa and the CIS is important to note. Similarly pig emission factor differences in rapidly industrializing developing regions such as Asia (particularly China) and

Table A3.2

Manure management methane emission factors (EF) for cattle (kilogram CH_4 per head per year) by production system and world region. Tier 2 based estimates compared to tier 1 emission factors

Region	Dairy cattle		Other cattle		Pigs	
	Weighted EF	Tier 1 EF	Weighted EF	Tier 1 EF	Weighted EF	Tier 1 EF
Sub-Saharan Africa	2.5	1	1.5	1	1.6	2
Asia excluding China and India	18.6	16	0.8	1	7.4	4-7
India	5.3	6	1.5	2	12.4	6
China	12.9	16	1.0	1	7.6	4-7
Central and South America	2.4	2	1.0	1	9.6	2
West Asia and North Africa	3.8	2	2.4	1	1.7	6
North America	51.0	54	9.5	2	22.7	14
OECD excluding North America	41.8	40	10.9	20	11.1	10
Eastern Europe and the CIS	13.7	6	9.1	4	2.8	4
Other developed	12.8	1	1.9	1	21.7	6

Source: Own calculations.

Latin America will induce differences between our quantification and existing ones.

3.4 Estimating water consumption for feed production

Generally the estimation of the amount of water consumed by a particular crop is done in a mechanistic manner, using a more or less sophisticated modelling approach. At the regional and global level these approaches are generally simple and, therefore, subject to strong assumptions. Chapagain and Hoekstra (2004) for example, estimating water footprints of nations, base their crop water use estimate on the method of Allen *et al*. (1998), multiplying a reference crop evapotranspiration with a crop coefficient.

Crop variety and climate are considered in the method of the latter author, but climate is not considered by Chapagain and Hoekstra (2004). It is supposed that adequate soil water is maintained by rainfall and/or irrigation so that it does not limit plant growth and crop yield. This leads to considerable overestimates in the warm and drier regions, which the authors claim to be compensated by their neglect of irrigation losses, but the latter locally unused portion of irrigation water is now widely acknowledged as not being lost at all (Molden and de Fraiture (2004).

For this report, we circumvented such problems by adopting a more deductive approach: detailed spatial information on arable land in general as well as separately for a number of important feedcrops has recently become available at the global level. This information was combined with spatially detailed and calibrated water balance and irrigation water-use estimates (FAO, 2003a: Box 4.3). The water balance calculation considers local precipitation, reference evapotranspiration, soil moisture storage properties, extents of areas under irrigation and irrigated areas for all major crops. Irrigation water consumption (in equipped areas) is calculated as the water required in addition to water from precipitation (including runoff from upstream areas) for optimal plant growth during the growing season.

This information avoids using statistics on water use or withdrawal, which would involve the difficult consideration of irrigation efficiency. At the same time the detailed information on the spatial distribution of important feed crops avoids having to combine the previous water consumption information with national level yield statistics, which would have been incompatible with assumptions of the water balance calculation.

One important difficulty remained though: before overlaying the overall crop maps with the irrigation and rainfed water consumption maps it needs to be determined in what locations crops are destined to be used for feed. Such information does not exist at the global level. However, we can assess the situation using two possible extreme hypotheses:

Hypothesis 1: *spatial concentration of feed*. Certain areas are entirely dedicated to the production of feed, and by matching their production with the national feed production statistics it is assumed that feed production elsewhere is insignificant.

Hypothesis 2: *area wide integration of feed.* Supposing a uniform distribution of both food and feed across cropland, it is assumed that anywhere where one of the considered crops is cultivated, a portion of the production equal to the national average is dedicated to feed.

In order to get an idea of the precision with which feed water consumption can be estimated, we used both approaches. A large difference between the two results would have suggested considerable uncertainty. The actual results (given in Chapter 4) show that the two approaches yield similar outcomes, indicating a degree of confidence in the results. Unfortunately, the detailed global crop maps are only available for a limited number of feedcrops. The crops considered in this assessment are barley, maize, wheat and soybean (hereafter called BMWS).

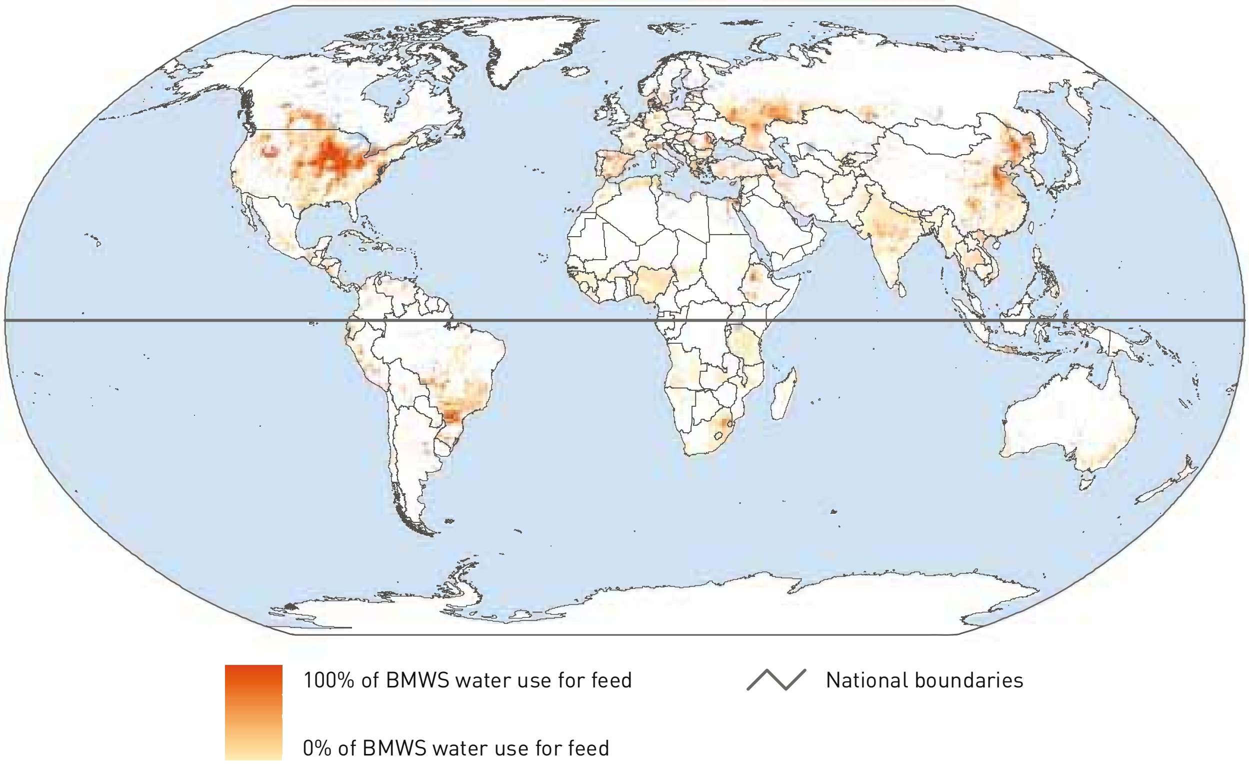
The area corresponding to hypothesis 1 is estimated in the following way: BMWS production dominates total local crop production. In addition, the combined production of barley, maize and soybean in this area is to be much larger than that of wheat (of which, generally, considerably less is used for feed). This latter criterion was used as an adjustable parameter to calibrate the area's size with respect to the national statistics on the barley, maize and soybean combined harvested area. Areas of BMWS production dominance were defined as those areas where the combined yield (making use of a recent detailed cropland area fraction map; exceeds 100 tonne/km^2. In the resulting areas country specific "aggregate" feed fractions are used to attribute water consumption in the area to the production of feed. This aggregate fraction is calculated as a weighted mean based on the production of barley, maize and soybean in the area and their corresponding national average feed use fractions (FAO, 2006b). In the particular case of soybean a fixed fraction of 66 percent was used, corresponding to the soymeal value fraction (Chapagain and Hoekstra, 2004).

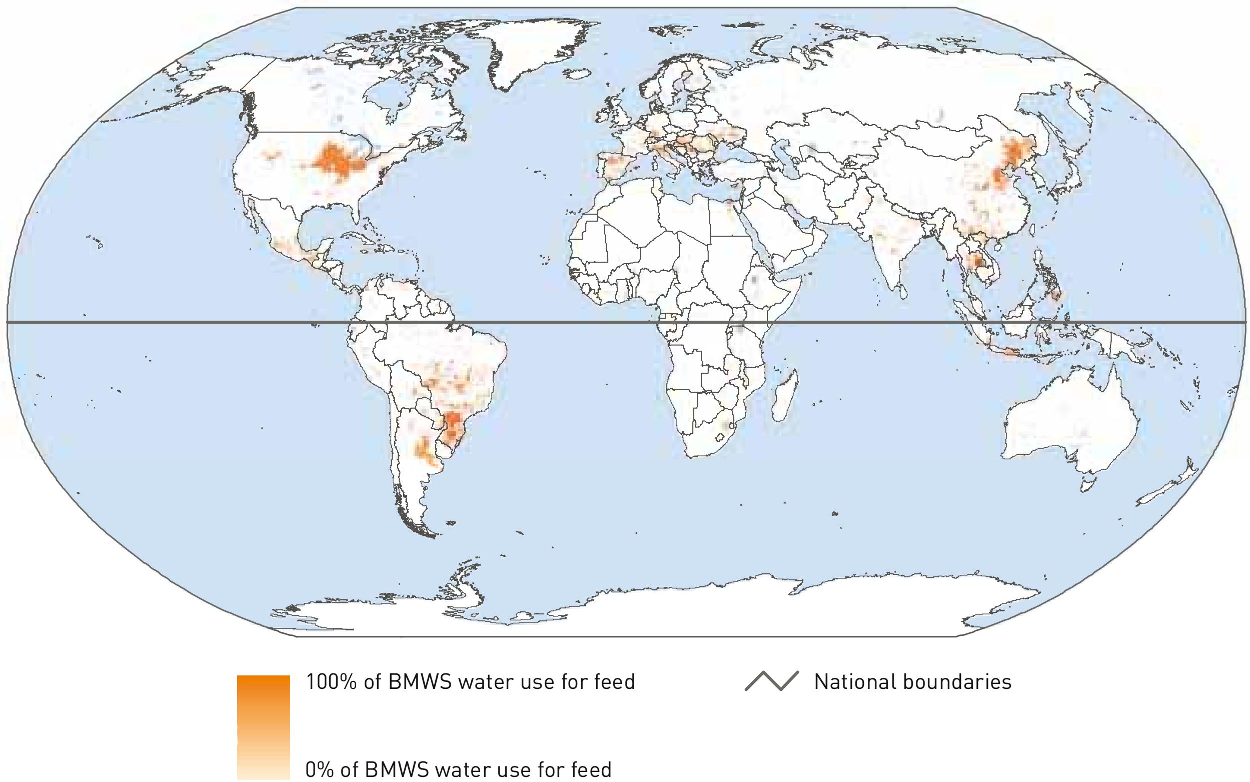
Under hypothesis 2 the entire BMWS area covered (as shown in the respective crop production maps) is considered to produce feed, but only to the extent corresponding to the national feed fraction of production (according to the supply utilization accounts in FAO). Again for soybean the 66 percent value fraction is used. Dividing the resulting summed BMWS feed production by the total summed BMWS production results in a map of local BMWS feed production fractions. In a final step towards determining the local fraction of water consumption for feed, the feed production fractions are multiplied by the BMWS cultivated fraction of the area (respective to other crops). These area fractions are defined as the sum of the individual crop areas (estimated by dividing production maps by national average yields) divided by the total cropped area.

The maps at the end of this annex show the contrasting feed production distributions resulting from both approaches. The apparent contrast in the corresponding water consumption is less dramatic than it seems, because different portions of consumption are attributed locally to the production of feed under the two hypotheses. These portions are generally higher under hypothesis 1 than under hypothesis 2.

The BMWS feed water consumption that

Map 1 **Water consumption for feed production: barley, maize, wheat and soybean**





Source: LEAD. Feed production areas (all non-white land area) and the feed fraction of BMWS water use resulting from Area Wide Integration of Feed approach (top map) and the Spatial Concentration of Feed approach (bottom map).

results from this assessment (Table 4.7) does not represent the entire water consumption for the production of feed. Figures 2.6 and 2.7 (Chapter 2) showed that these four crops together constitute about three quarters of feed concentrates for pig and chicken, i.e. the global total water consumption for feed may roughly correspond to 1.3 times that of BMWS feed. Finally, it is worth emphasizing that these estimates exclude water consumed for the production of natural but grazed grass and cultivated fodder. Their inclusion would substantially change the feed water consumption estimates, particularly on the rainfed consumption side. However, much of the consumption of grazed grass does not have an opportunity cost as is the case for cultivated areas. Including this, if it had been possible would, therefore, reduce the environmental relevance of the result.