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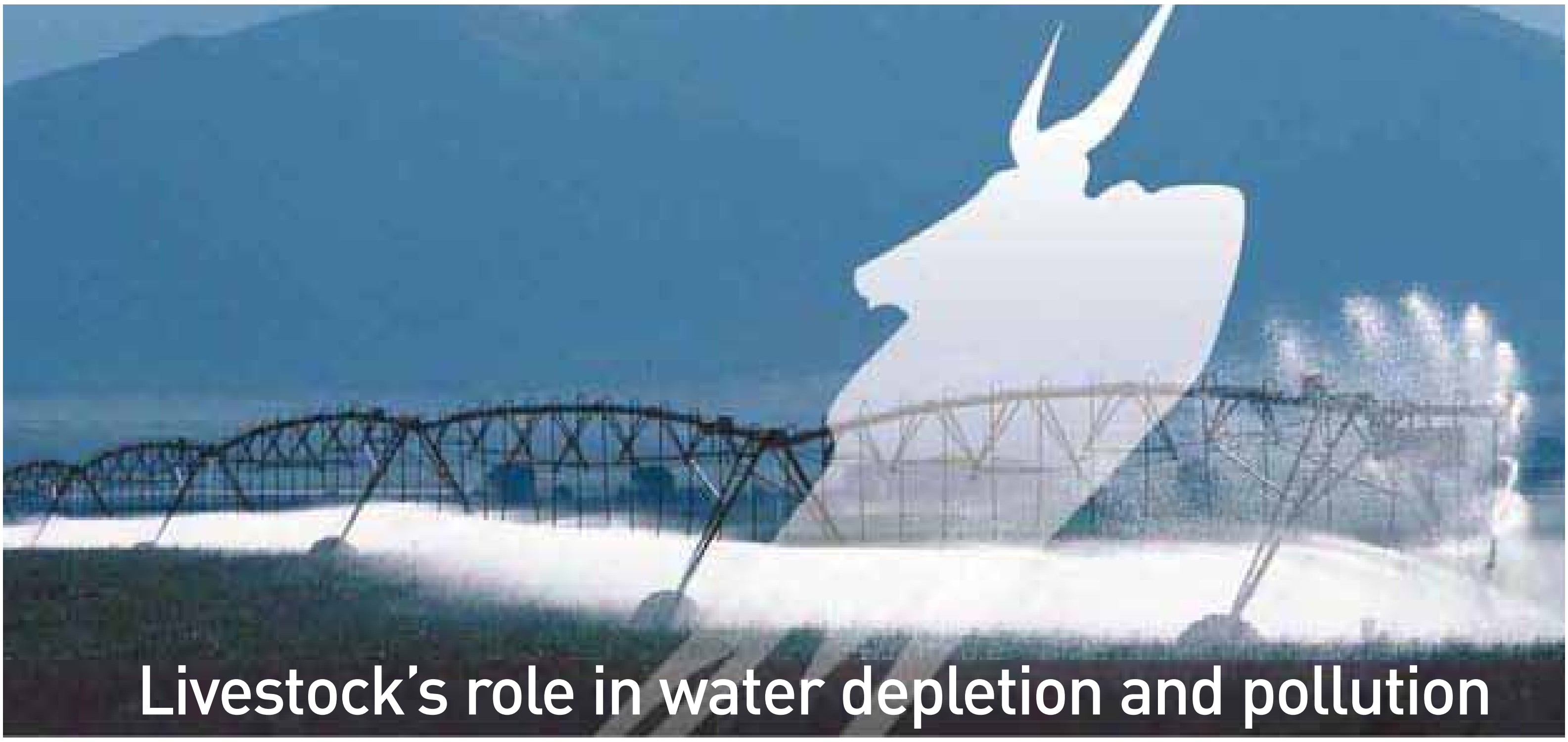

Livestock's role in water depletion and pollution

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.

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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

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[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

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[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

Animal	Age group	Service water *(litres/animal/day)* 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.

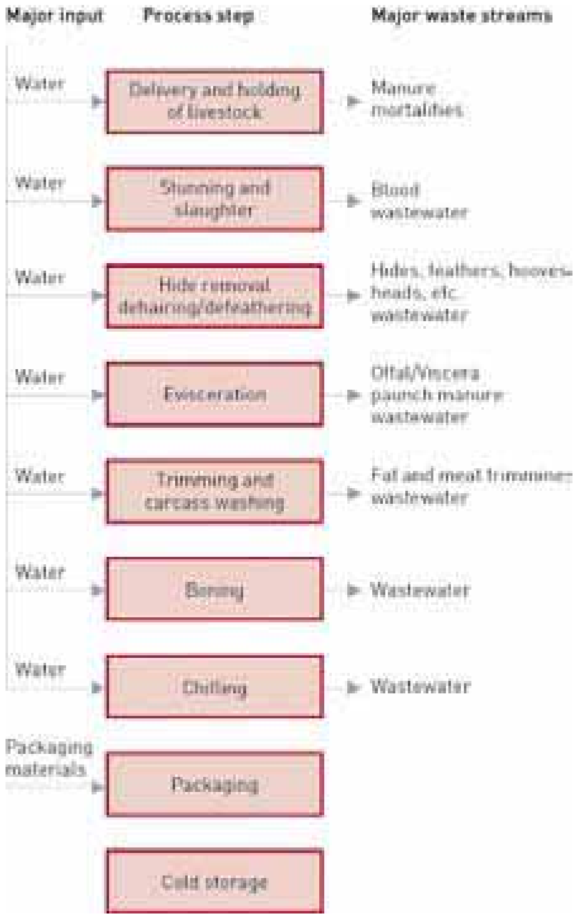
Tanneries

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).

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)

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 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

© PHOTO COURTESY OF USDA NRCS/CHARMA COMER

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

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[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-

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[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

	Irrigated BMWS feed			Rainfed BMWS feed		BMWS feed irrigation water ET as percentage of total BMWS feed water ET
Region/Country	Evapotranspired irrigation water km^3	Percentage of total irrigation water evapotranspired	Percentage of total water evapotranspired in irrigated areas[1]	Water evapotranspired km^3	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 ($N0_3$-) (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 administered 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

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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).

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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	49
Eastern Africa	41.0	109.0	148.0	76.0	24.0	31.0	78.0	40.0	
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	10
East Asia	95.0	24 345.0	5 150.0	3 358.3	29.0	0.0	1 404.0	915.5	
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).

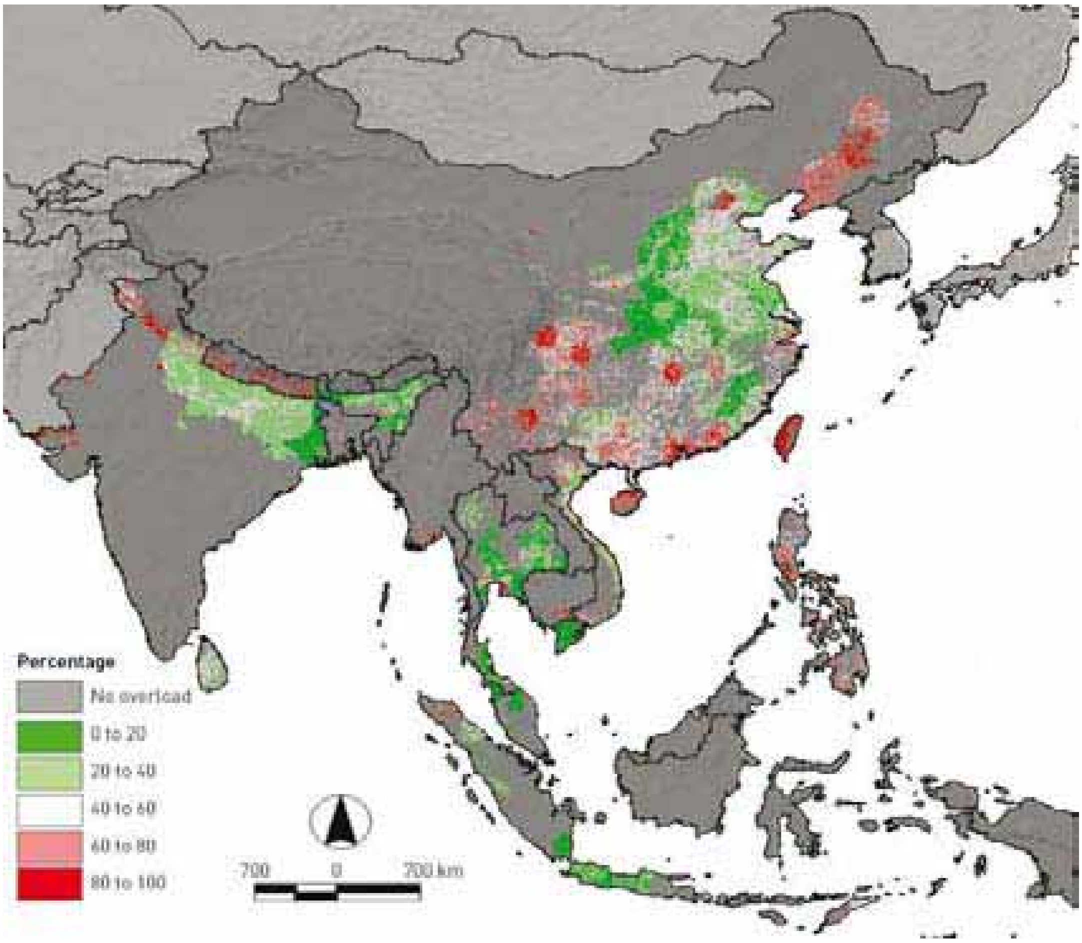
Looking at the European target values for urban waste discharge (e.g. 25 mg BOD, 1 015 mg N and 12 mg P per litre), wastewater from slaughterhouses has a high potential for water pollution even when discharged at low levels. Indeed, if directly discharged into a water course, the wastewater originating 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

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[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		Livestock contribution	
	N	P	N losses	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

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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:

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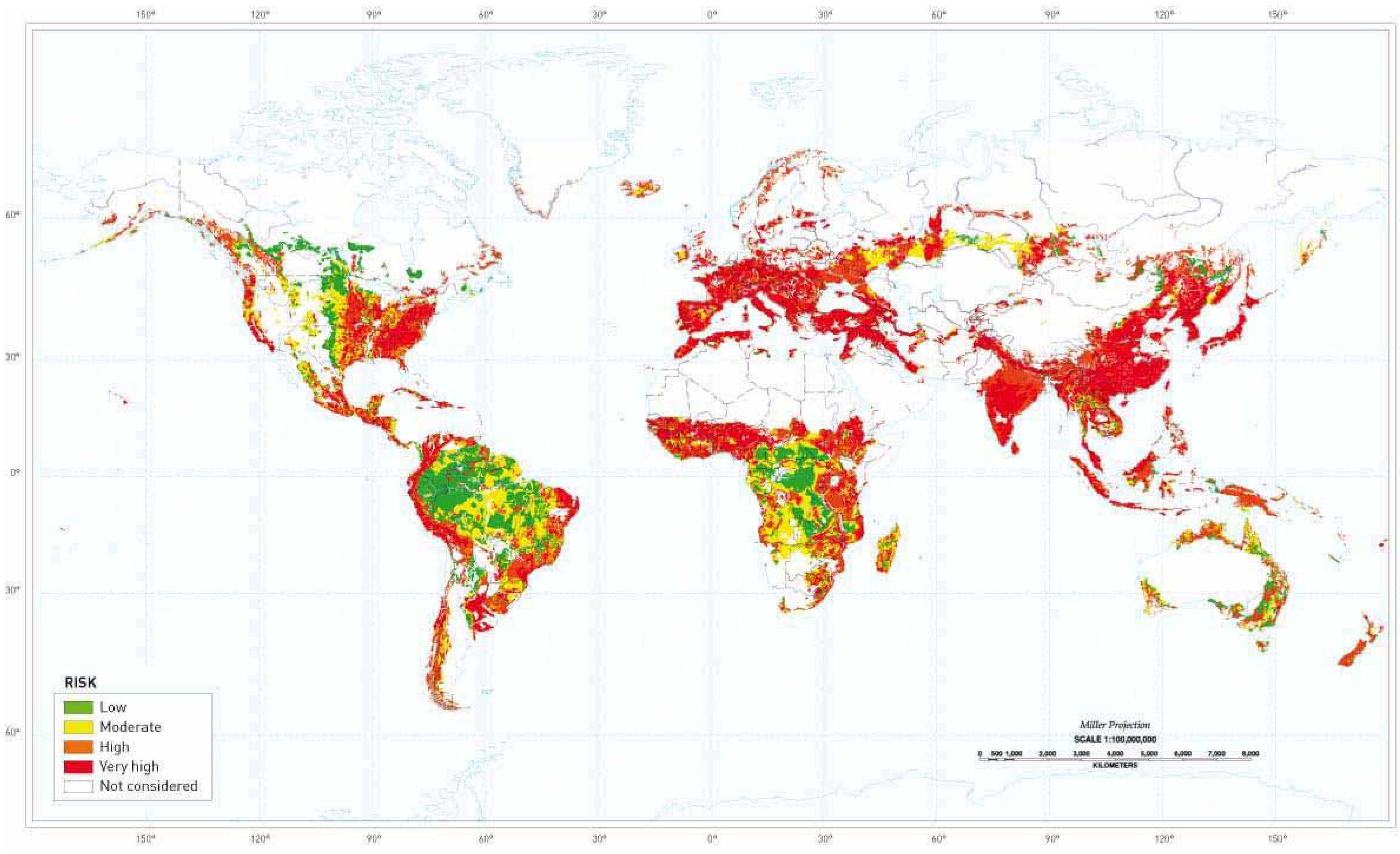
River bank soils loosened by water buffaloes in Naning, China causing sedimentation and turbidity

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-

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[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.

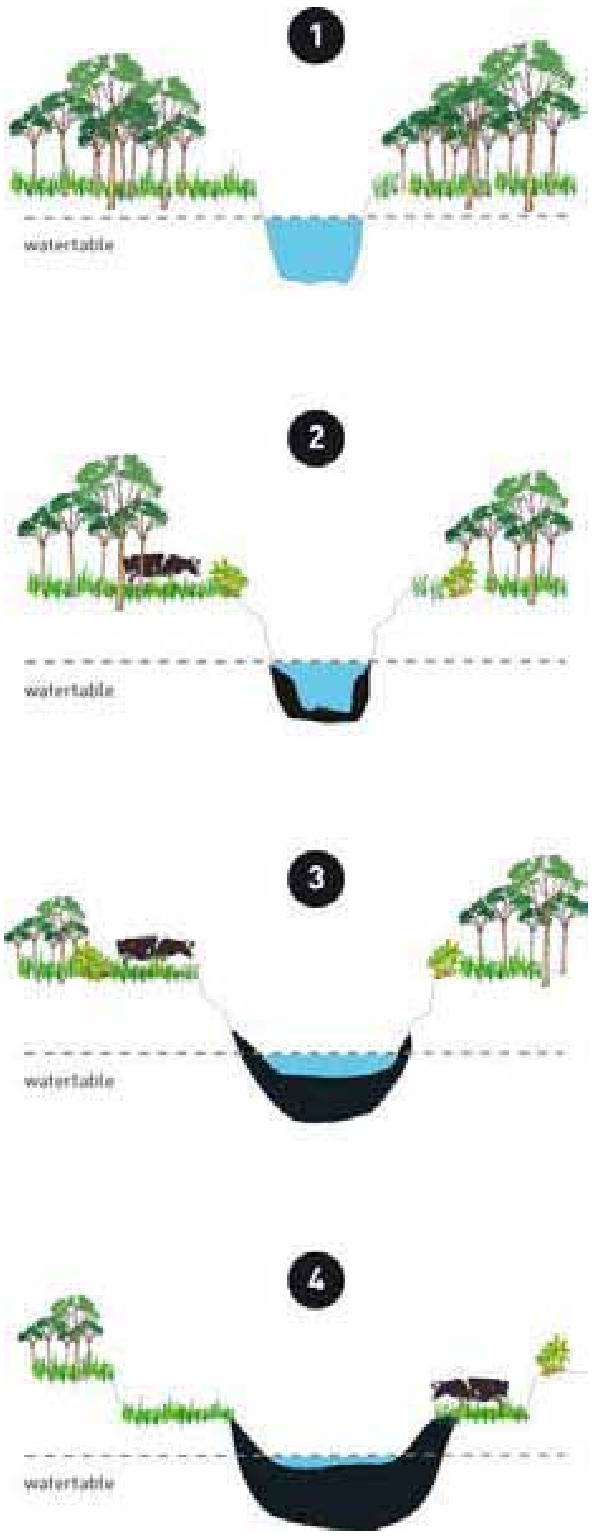
Grazing animals are also important agents of geomorphological change as their hooves physically reshape the land. In the case of cattle, the force is usually calculated as the mass of the cow (500 kg approx.) divided by the basal hoof area (10 cm^2). However, this approach may lead to underestimates, 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

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[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

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[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

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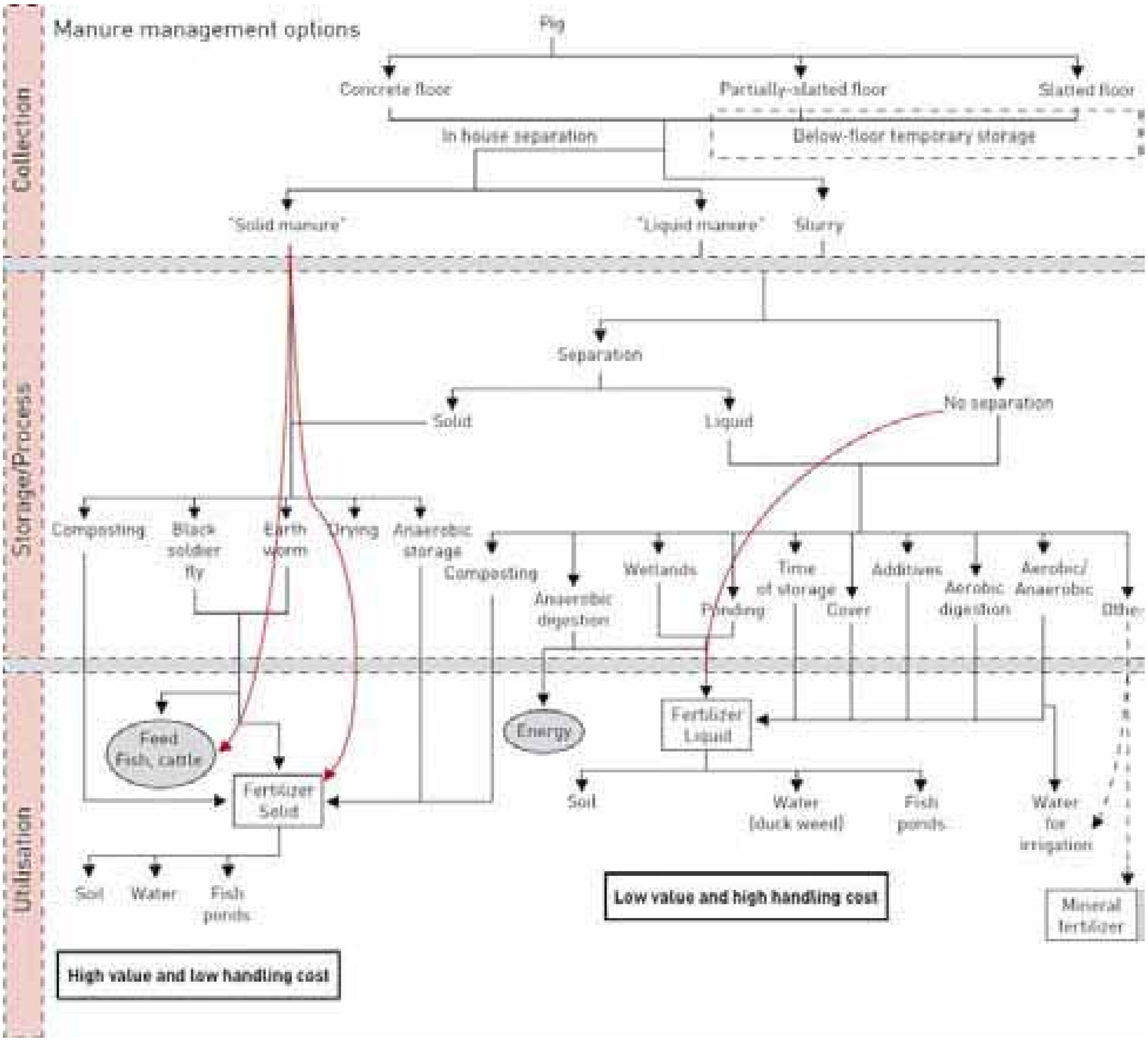
[11] The activated sludge process uses the organic matter of wastewater to produce a mixed population of microorganisms, in an aerobic environment.

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.