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Guidelines for Sustainable Manure Management in Asian Livestock Production Systems

A publication prepared under the framework of the RCA project on Integrated Approach for Improving Livestock Production Using Indigenous Resources and Conserving the Environment

Joint FAO/IAEA Programme
Nuclear Techniques in Food and Agriculture

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FOREWORD

The International Atomic Energy Agency (IAEA) and the Regional Cooperative Agreement for Asia and the Pacific Region (RCA), with the technical support of the Joint FAO/IAEA Programme of Nuclear Techniques in Food and Agriculture, implemented a Technical Cooperation (TC) project entitled “Integrated approach for improving livestock production using indigenous resources and conserving the environment” (RAS/5/044). Technical Cooperation projects are technology transfer initiatives, designed to address specific priorities identified by Member States. The specific objectives of this project were: (a) to improve animal productivity and decrease discharges of selected greenhouse gases, (methane and carbon dioxide) and selected nutrients (nitrogen and phosphorus) into the environment; and (b) to identify and adopt better breeding strategies to improve animal productivity through the use of better selection criteria for offspring from cross-breeding programmes, optimum utilization of appropriate indigenous cows, benchmarking for growth and reproduction, and improving procedures for management, nutrition and healthcare programmes in dairy farms.

The first meeting to plan project activities was hosted by the Institute of Agricultural Environment and Sustainable Development of the Chinese Academy of Agricultural Sciences (CAAS), Beijing, and was held from 4 to 8 April 2005. It was attended by 23 nominated project counterparts from 12 RCA Member States and was supported by three IAEA experts. One of the conclusions from this meeting was that there was considerable scope and need for improving current manure management practices in the region to enhance the productive recycling of ingested nutrients in animal production systems, which in addition to increasing livestock and crop productivity will decrease environment pollution. It was agreed that there was a need to focus on improving the nutritional and manure management in integrated livestock systems, and that it was important to evaluate alternative management strategies on-farm. It was recommended that a consultants meeting on manure management should be held in 2005 to review current manure management practices in the region and to develop guidelines for efficient management of manure in different livestock production systems. It was also recommended that these guidelines be distributed to the groups participating in RAS/5/044 so that they could form the basis for planning manure management work to be conducted during the second phase of the project.

To address these recommendations, an experts meeting on ‘Development of guidelines for efficient manure management in Asian livestock production systems’ was organized and held in Ho Chi Minh City, Vietnam. The meeting was hosted by the Institute of Agricultural Science (IAS) for Southern Vietnam and was held from 5 to 9 December 2005. It was attended by 7 foreign experts and one local expert, and was supported by the Technical Officer (TO) of RAS/5/044. The experts gave presentations on state-of-the-art manure management practices and participated in a field visit to experience first hand some of the current manure management practices in Asia. After in depth discussions about the presentations, taking into account the experiences of the field visit, and identifying the target audience for guidelines of this type, an outline of the guidelines for manure management was developed, including recommended or ‘best practice’ manure management strategies and an inventory of available materials on the subject. What was clear from the discussions and planning was that there is very little information and few guidelines about manure management for Asian livestock systems.

A draft document was prepared during the meeting but because of the lack of information available on manure management in Asia, there was not enough time to complete a final draft of the guidelines at the meeting. H. van der Meer kindly took on the responsibility of editing the chapters and ensuring that the final document of the guidelines was completed. This was a major undertaking because it involved an extensive review of the literature for manure management strategies and guidelines that were relevant and could be applied to livestock production systems in Asia. The contribution of H. van der Meer in preparing this report, and the financial support of the Netherlands Ministry of Agriculture, Nature and Food Quality and the Agrosystems Research Business Unit of Plant Research International at Wageningen for H. van der Meer to undertake this task, are gratefully acknowledged. A list of contributors to this publication is presented at the end of this report, and their contribution is also thankfully acknowledged.

This manual includes information about trends in livestock production and animal manure management in Asia, systems approach to sustainable manure management, production and composition of manure, manure management during housing and storage, processing and handling of manure to reduce pollution and improve nutrient utilization, field application and utilization of manures, and the main conclusions and recommendations from the experts meeting.

This manual is aimed at all levels of administrative and technical personnel involved in the management of manure in livestock systems and environmental sustainability in Asia, including Ministries of Agriculture/Livestock/Environment, Directorates of Livestock and Veterinary Services, local authorities responsible for livestock development services, Faculties of Agriculture and Animal/Plant/Soil Sciences, and Institutions involved in environmental sustainability. It is hoped that the manual will assist livestock personnel in Asia to apply the guidelines to improve existing management systems and develop new systems that are efficient, cost effective and sustainable for different livestock farming systems under varying socio–economic environments.

The IAEA officers responsible for this publication were P. Vercoe, P. Boettcher and H. Makkar of the Animal Production and Health Section of the Joint FAO/IAEA Programme of Nuclear Techniques in Food and Agriculture.

CONTENTS

1. INTRODUCTION

Livestock manures represent a valuable resource that, if used appropriately, can replace significant amounts of chemical fertilizers [1, 2]. However, unless animal manure is managed carefully to minimize odour, nutrient losses and emissions, it becomes a source of pollution and a threat to aquifers and surface waters [3]. It can also be a direct threat to human and livestock health. Animal production is developing rapidly in Asia and the impact of mismanaging manure could be detrimental. There is an urgent need to review and assess current manure management practices and develop manure management guidelines that are appropriate for adoption by local animal producers.

Mismanagement of manure often leads to direct discharge of liquid manure to waterways. This causes serious eutrophication of rivers and lakes, characterized by a high concentration of nutrients that creates an ecological imbalance in the water system because it supports abnormally high levels of growth of algae and aquatic plants, *e.g.* water hyacinths. This decreases oxygen levels in the water and has serious implications on the survival of other organisms in the system and, consequently, on food supply and biodiversity. Furthermore, surface- and ground-waters can be polluted by leaching and run-off of manure nutrients and this increases the need for water purification treatment to provide safe drinking water.

Direct discharge of manure to waterways and percolation to groundwater, usually in by-pass flow *via* cracks and fissures, is a great risk to human and animal health because livestock manure contains numerous pathogens (bacteria, viruses, parasites). Some of these may be transmitted to man, and can cause systemic or local infections, *e.g. Escherichia coli*, *Campylobacter*, *Salmonella, Leptospira, Listeria, Shigella, Cryptosporidum, Hepatitis A, Rotavirus, Nipah virus, Avian Influenza* [4–9]. The annual occurrence of typhoid fever has been estimated at 17 million cases with approximately 600,000 deaths, and diarrhoeal diseases cause death of 951,000 people annually in South East Asia [10]. A number of water-borne bacteria, protozoa, viruses and, in particular, parasites are the source of these diseases. Transmission of these pathogens is enhanced by inappropriate management of animal manure and may be reduced by proper manure handling and use. Insanitary handling of manure may also promote the spreading of parasites to man by introducing larval stages into the food chain. Recently, reviews on the current status of parasitic diseases in Vietnam, including consideration of food-borne trematode zoonoses and cysticercosis have highlighted the risks of disease transmission through animal manure and human excreta [11, 12]. Highly contagious and pathogenic diseases, such as Foot and Mouth Disease, Swine Fever and Aujezsky's Disease may also spread with animal effluent through waterways and, when one farm is infected with the disease, farms downstream will be at considerable risk of infection [9]. It has not been completely proven, but poor manure management, the mixing of human and animal excreta, and the close contact between domestic and animal housing may propagate Avian Influenza [10] and Severe Acute Respiratory Syndrome.

In contrast, careful recycling of animal manure to land will contribute plant nutrients to crops and reduce the need for mineral fertilizers. In animal manures, nitrogen (N) is in both an inorganic (ammonium) and organic form, whereas phosphorus (P) and potassium (K) have a fertilizer value equivalent to that of mineral fertilizers [13]. The N fertilizer value of manure in the first growing season is lower and more variable than that of commercial fertilizers, because a variable part of the inorganic N is lost by ammonia (NH_3) volatilization [14], depending on the rate and period of application, weather conditions and soil type, run-off, denitrification and leaching [15–18]. Manure organic N has to be mineralized before it is available to plants [19–21]. However, the value of the manure as fertilizer can be increased substantially provided there are guidelines for using animal manures.

Livestock production units are also a source of malodours originating from livestock buildings, and storage and field application of animal manures. The intensity of malodours is often unacceptable, especially for neighbours in surrounding residential areas, and studies indicate that pig production units have a negative impact on the sale value of nearby dwellings. Globally, the concentration of the greenhouse gas methane (CH_4) in the atmosphere has increased by 45% since 1850 [22]. Increases in livestock production have contributed significantly to this increase and it has been estimated that enteric fermentation of ruminants contributes some 13–15% and livestock manure 5% to the total emission of CH_4 in the 1990s [22, 23]. The emission of nitrous oxide (N_2O), a very potent greenhouse gas, has increased from 11 Tg year^{-1} in 1850 to 18 Tg year^{-1} in mid 1990s, mainly due to increases in anthropogenic sources. Agriculture was estimated to have contributed almost 80% to the anthropogenic emissions of N_2O in the 1990s [24]. Further, emission inventories show that livestock production contributes 70–80% of the anthropogenic NH_3 emission in Denmark and Europe [25, 26]

Animal production is developing rapidly in Asia. The trends towards specialization and intensification on larger production units to improve profitability have resulted in the pollution of air, water and soil [27, 28]. There is a pressing need for holistic research into strategies and technology for management and treatment of manures, which can ensure a sustainable use of nutrients and mitigation of environmental impacts, including odour and NH_3 emissions, greenhouse gas emissions and the spread of diseases (Figure 1.1).

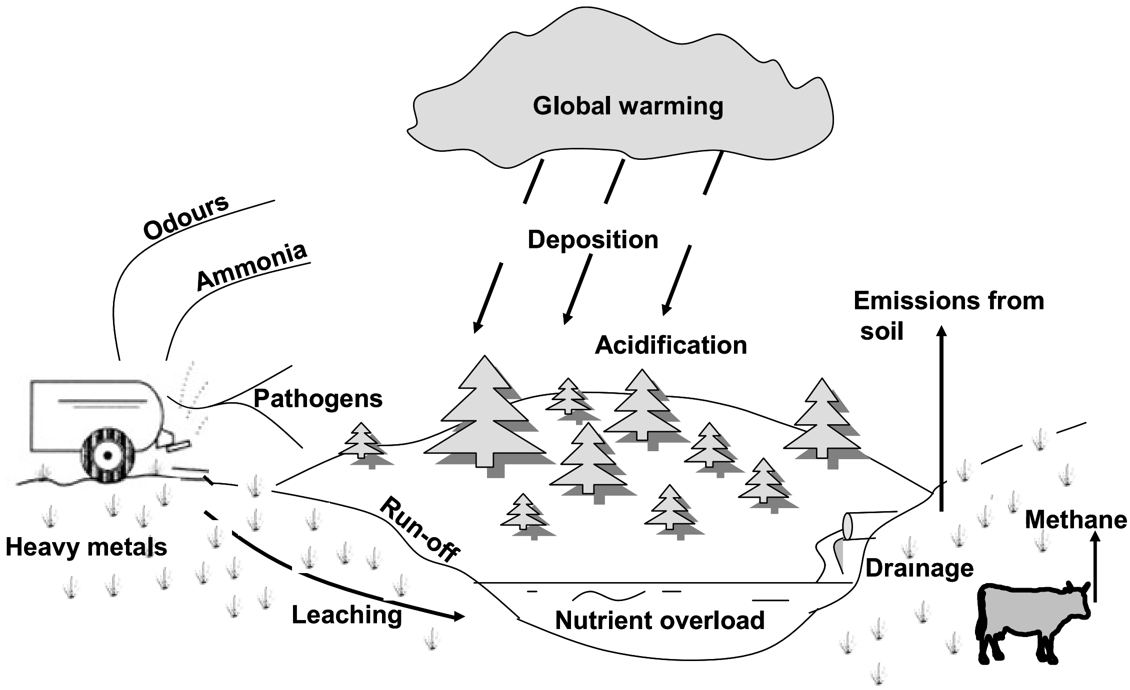

FIG. 1.1. Environmental hazards related to the management of animal manures [29].

In this report, we have developed guidelines that address the benefits of using manure in plant production and the consequences of its mismanagement. These guidelines, therefore, address different aspects of manure production, collection, storage and processing, and utilization. The guidelines may be further developed and improved in Asia by quantitative studies of N and P flows related to manure production, handling and field application [30]. This should give farmers confidence in the reliability of using manure as a fertilizer and contribute towards the development of sustainable, environmentally friendly livestock production systems in Asia, and with reduced risks of disease transmission.

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2. TRENDS IN LIVESTOCK PRODUCTION AND ANIMAL MANURE MANAGEMENT IN ASIA

2.1. TRENDS IN LIVESTOCK PRODUCTION IN ASIA

Livestock production in East, South and Southeast Asia increased rapidly during the last decades of the 20th century and it is the policy of many countries to further increase meat and milk production [1–5]. The term 'Livestock Revolution' has been used to describe this development. The growth of meat and milk production is driven by the growth in demand, and this is fuelled by increasing incomes, urbanization and changing lifestyles, and population growth. The trends in per capita meat and milk consumption in Asian regions and, for comparison, in the whole developing world as well as in the developed world are presented in Table 2.1.

TABLE 2.1. PER CAPITA CONSUMPTION OF MEAT AND MILK IN ASIAN REGIONS IN 1983, 1993 AND 2020 (PROJECTED). SOURCE: COMPILED FROM [5]. THE VALUES ARE THREE-YEAR MOVING AVERAGES CENTERED ON THE YEARS SHOWN

Region	Per capita meat consumption (kg)[1]			Per capita milk consumption (kg)[2]		
	1983	1993	2020	1983	1993	2020
China	16	33	60	3	7	12
Other East Asia[3]	22	44	67	15	16	20
India	4	4	6	46	58	125
Other South Asia[4]	6	7	10	47	58	82
Southeast Asia[5]	11	15	24	10	11	16
Developing world	14	21	30	35	40	62
Developed world	74	76	83	195	192	189

[1] Meat includes beef, pork, mutton, goat, and poultry carcass weights; [2] Milk is cow and buffalo milk and milk products in liquid milk equivalents; [3] Hong Kong, Macau, Mongolia, North Korea, South Korea; [4] Afghanistan, Bangladesh, Bhutan, Maldives, Nepal, Pakistan, Sri Lanka; [5] Brunei, Cambodia, East Timor, Indonesia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, Vietnam

Growth rates of meat consumption are particularly high in China, Other East Asia and Southeast Asia, whereas those of milk consumption are high in India, Other South Asia (in particular Pakistan) and China (Table 2.1). In 2003, per capita meat consumption in China was estimated at 55 kg [5]. This means that the average rate of increase between 1983 and 2003 was almost 2 kg per capita per year. In other countries with fast growing meat consumption, like Malaysia, Philippines, Republic of Korea, Thailand and Vietnam, the rate of growth has been *ca*. 1 kg per capita per year in the same period. (NB. Many reports present the rate of increase of consumption or production as an annual percentage. However, this is misleading, because it suggests exponential growth over a longer period and this does not agree with reality). A rapid increase in the consumption of milk and dairy products has been observed in India and Pakistan. Since 1983, per capita consumption of fresh and manufactured milk in India and Pakistan increased at rates of *ca*. 1.5 and 3 kg fresh milk equivalent per year, respectively [5]. Half of the present consumption of milk in these countries is in a manufactured form.

Per capita growth of meat and milk consumption in combination with population growth caused a very strong increase in meat and milk production in many Asian countries (Table 2.2).

TABLE 2.2. DEVELOPMENT OF TOTAL MEAT AND MILK PRODUCTION IN ASIAN COUNTRIES AND, FOR COMPARISON, IN THE WHOLE DEVELOPING WORLD AS WELL AS IN THE DEVELOPED WORLD. SOURCES: COMPILED FROM [5] (YEARS 1983, 1993 AND 2004) AND [1] (PROJECTION 2020). FIGURES ARE THREE-YEAR MOVING AVERAGES CENTERED ON THE YEARS INDICATED

Region	Total meat production (million metric tons)				Total milk production (million metric tons)			
	1983	1993	2004	2020	1983	1993	2004	2020
China	17	41	74	86	4.0	8.3	26	19
Other East Asia	1.1	1.8	2.2	7	1.0	2.2	2.8	3
India	3.0	4.4	6.1	8	39	59	91	172
Other South Asia	1.3	2.3	2.8	4	12	20	33	46
Southeast Asia	4.5	7.8	11.4	16	1.0	1.5	2.6	3
Developing world	53	93	150	183	124	176	269	401
Developed world	92	102	109	121	370	353	354	371

See notes in Table 2.1.

Since 1995, meat production in the developing countries exceeded that of developed countries (Table 2.2). Total meat supply in the developing countries has almost tripled from 53 million metric tons in 1983 to 150 million metric tons in 2004. China alone accounted for almost 60% of this increase. Globally, China is the number one producer of pork, mutton and eggs with outputs in 2004 of 46.7, 3.6 and 28.1 million metric tons, respectively (Steinfeld & Chilonda, 2006). The number of pigs slaughtered in China was *ca*. 650 million in 2005 [5]. A rapid increase in meat production is also observed in Southeast Asia (Table 2.2).

Growth of milk production in developing countries has been less spectacular than that of meat production (Table 2.2). However, total supply has expanded from 124 million metric tons in 1983 to 269 million metric tons in 2004. Recently, India has emerged as the number one producer of milk and dairy products in the world, with an estimated output in 2004 of 91 million metric tons [2]. Further growth of milk production in India and Other South Asia will depend strongly on the availability of good forages and concentrate ingredients and this point questions the very high projection for 2020.

Simultaneously with the increase in livestock production in Asia, production patterns have changed and more intensive or industrial livestock production systems have emerged [6]. At present, a large proportion of livestock is kept for food production and the traditional functions of providing draught power and manure and serving as a capital asset are becoming less important. This also means a shift from livestock in the back-yard of small farms as converters of household residues and low-quality forages to livestock in specialized production units fed home-grown or purchased feedstuffs. In China, this intensification of livestock production started in the 1970s and has been closely related to the rise in income since then [7]. The increased demand for fresh meat and milk within prospering urban centers and the lack of efficient infrastructure in rural areas have resulted in a large concentration of livestock production near cities [3]. In a subsequent phase of economic growth and industrialization, infrastructure and technology will need to develop sufficiently to make it possible to keep livestock at greater distances from centers of demand. Environmental pollution and risks for human health may generate pressure to relocate livestock production in rural areas [8]. This may also be driven by factors such as lower land and labour prices, better access to feed, lower environmental standards, tax incentives, and fewer disease problems [9].

In many developing countries in Asia, the majority of the population still finds employment in agriculture. For instance in Vietnam, agriculture still provides an income for approximately 65% of the population and although this percentage is decreasing, the absolute number of people economically active in agriculture is expected to increase in the next years [5, 10]. Recently, *ca*. 85% of the animals were produced by small holders [11]. Increasing animal production on small farms offers a good chance to earn additional farm income. However, several developments have stimulated a move away from small holder operations towards an increase in meat and milk production on larger commercial farms [9]. For instance, large supermarkets in countries such as China demand milk of a high hygienic standard. Machine milking is considered to provide a better hygienic standard and this is expected to further intensify the shift from small dairy units to large ones. However, while developing technology for efficient and environmentally friendly production systems, one should consider not only intensive livestock production systems but also small-holder and medium-scale livestock producers. The small farms may comprise up to, say, 10 fattening pigs, 100–200 birds, or 5 dairy cows. Furthermore, it should be noted that the specialization in livestock production tends to weaken the linkage between livestock and plant production. Consequently, an increasing number of intensive livestock production units of different size, with insufficient land for a sustainable recycling of manure nutrients, are emerging.

2.2. HOUSING SYSTEMS

2.2.1. Small livestock producers

In most situations, pigs are kept in houses with solid floors. The farmers often separate excreta into liquid and solid fractions manually (Figure 2.1). Excreta are scraped off the floor, composted and used for vegetables and fruit crops. Urine and some solids are washed from the floor and, subsequently, discharged to fish ponds or surface water [11]. On a few small farms, the fattening pens are divided into two parts, a living area with concrete floor and a lower lying channel area where excreta are collected by scraping the solid floor area and mixed with straw. Many farms have small biogas plants producing a gas for cooking from the liquid manure.

Buffaloes are housed during the night. During periods when crops are growing in the fields, the children will herd the buffaloes for grazing along the dikes. After harvest the buffaloes graze on the fields. In the small pens the buffaloes are housed on concrete or earth floors with some rice or wheat straw bedding. In general, urine is discharged from the buffalo pens indicating that there is not enough straw to retain all urine excreted. In a Vietnamese village, it has been observed that liquid manure from all barns is transferred into a few fish ponds and as the nutrient content of the manure is neither known nor taken into account, this has caused eutrophication of the ponds and killed the fish.

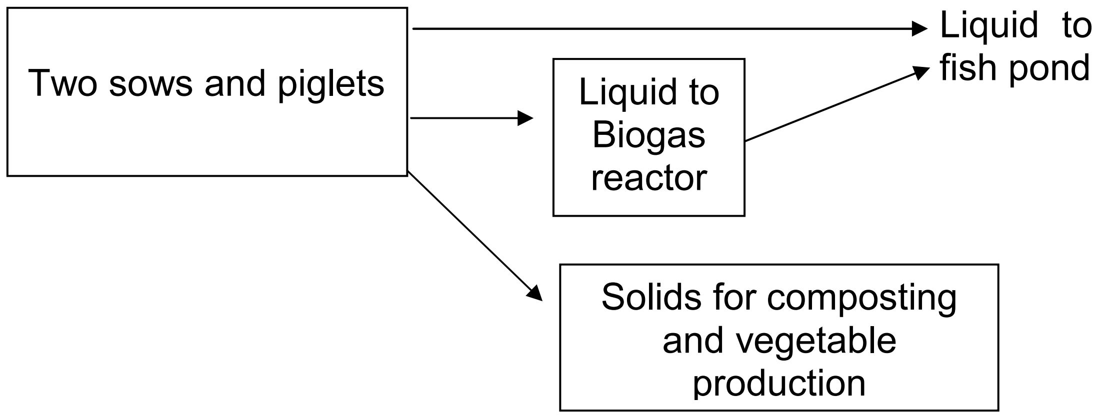

FIG. 2.1. Example of manure handling on a small farm in the Red River district near Hanoi.

2.2.2. Medium-size farms

These farms have tended to specialize in one type of production, *e.g.* dairy, pigs or poultry. In Vietnam, pig rearing on medium-size farms have been observed on steel slats raised 40–50 cm above the concrete floor (Figure 2.2). Manure solids are collected from both the slats and the concrete floor by scraping. In Malaysia, the pigs are traditionally reared on a solid concrete floor with the solids scraped and collected and the liquids being drained to anaerobic treatment ponds and, from there, discharged to rivers.

Dairy cows and buffaloes are kept in houses with concrete or earth floors with straw bedding. The concrete floors are cleaned by water and the animals cooled by hosing in some cases. In Vietnam, liquid manure may be discharged to fish ponds on some farms. Other farmers in Vietnam and all pig farmers visited in Malaysia were observed to discharge the liquid manure to canals or rivers after a short period of treatment in lagoons.

Poultry farmers raise the birds in open houses on wire or slatted floors and the manure is collected below the cages. In most Asian countries, chicken manure as well as solid manure of other animal categories is considered a commodity, valued US$ 5–10 per ton.

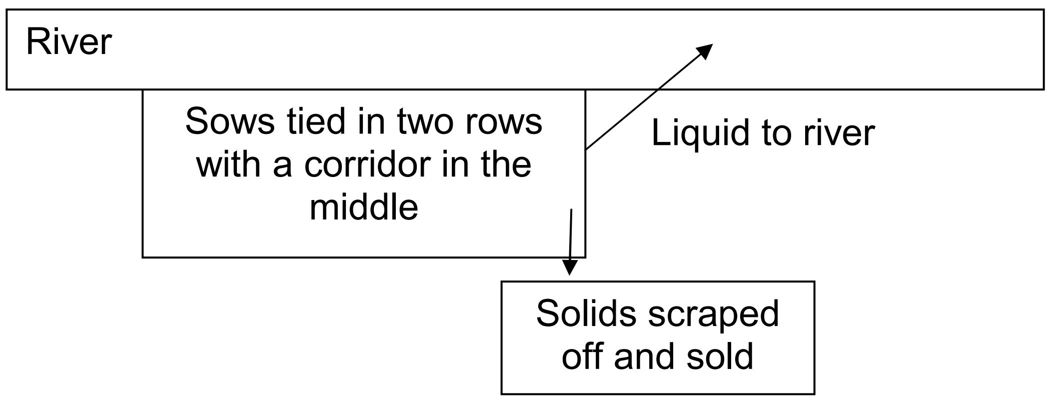

FIG. 2.2. Manure handling on a medium-size pig farm in Vietnam.

2.2.3. Large farms

Pigs are generally housed on solid or slatted floors (Figure 2.3). On farms visited in Vietnam and Thailand, the solids were collected by scraping the floor of the houses once or twice a day. The pens are then cleaned by hosing. On a large pig farm in Malaysia the solids were not separated from the liquid manure and a slurry was produced, which was separated by mechanical sedimentation. The dairy buildings and pig houses are cleaned by water and the animals, in some cases, cooled by hosing. Alternatively, the pigs are cooled by sprinklers in the house, by pouring water on the roof or drawing air into the house through a curtain of water in the gable end of the animal house. In Thailand, occasionally it has been observed that while systems have been designed with solids-liquids separation, the separator was not always running. On farms in Thailand and Malaysia the liquid manure was treated in aerobic/anaerobic treatment plant before being discharged to rivers. The passage of effluent through two or more lagoons in series reduces the discharge of nutrients and pollutants to rivers. The need for discharge is partly due to the lack of scope for using liquid manure (waste water) because fields are too far from the production unit and partly because farmers are not aware of the fertilizer value of the manure. Further, farmers may have problems in using the manure in a timely and proper manner; therefore the crop may be 'damaged' due to untimely application or nutrient oversupply.

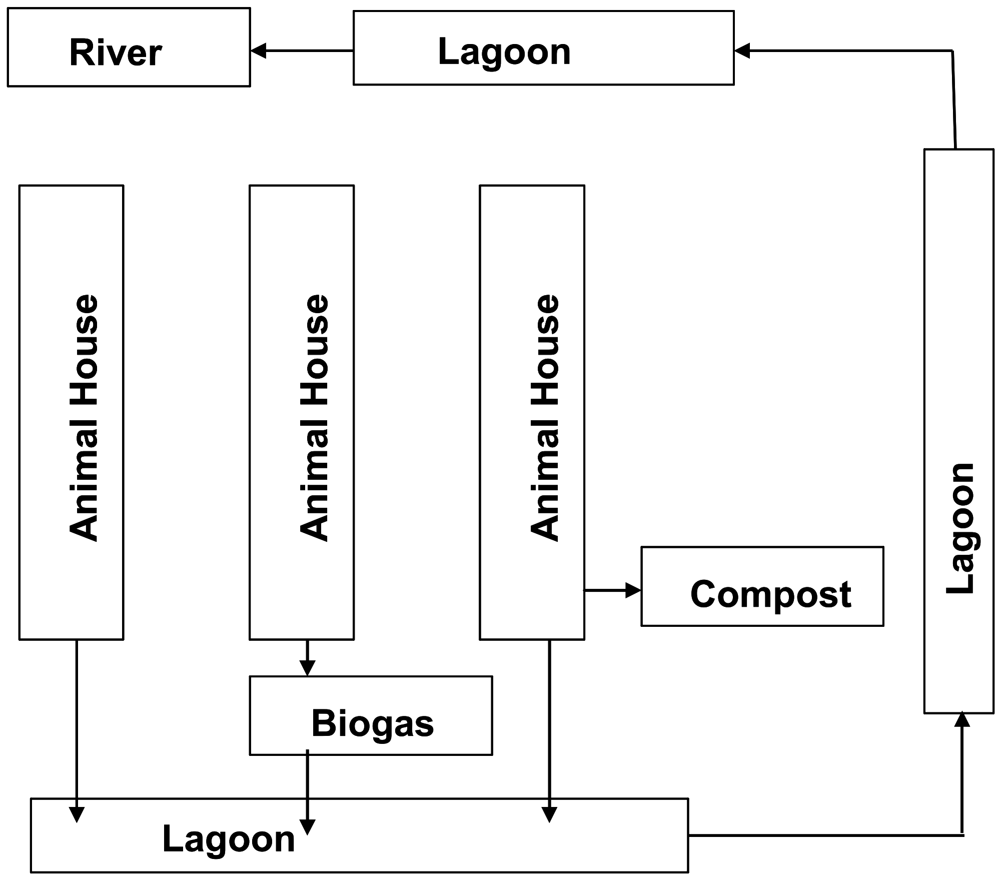

FIG. 2.3. Conceptual presentation of a large pig farm in Vietnam, Thailand or Malaysia.

2.3. MANURE TREATMENT IN ASIA

In Asia, little information is available about manure production per animal and composition of the manure as related to animal species and categories, production level and feeding practice. At present, the best data of this type appear to be based on various assumptions [3]. The loss of nutrients from agricultural operations will become a critical issue in areas that are exposed to eutrophication [12]. Further, there is a lack of regional specific data about losses of nutrients due to direct discharge, run-off and ammonia (NH_3) volatilization during manure handling, all of which constitutes a loss of fertilizer value and compromise the determination of correct field application rates of manures used as nutrient sources for crop production.

Plant nutrients may be lost due to leaching from solid manure heaps and from liquid manure stored in unlined, earth lagoons or run-off from the transport channels and lagoons during rain. The magnitude of these problems has not been studied adequately in Asia and on-farm research is needed both to assess manure production across the range of systems and to evaluate the flow of manure nutrients from excretion to use (crops, aquaculture). Further, there is a need to develop collection, handling, storage and transport systems that are compatible with the requirements of current production systems and farmer preferences, and are also environmentally sustainable. Solid and liquid manure is primarily stored to ensure a timely application of the manure to the crop and to reduce the viability of pathogens in the manure.

Fermentation of manure in biogas digesters is a well known technology in Vietnam [13] and in many other parts of Asia, but it has been observed in Thailand and Vietnam that gas production is low due to the low dry matter content of the liquid fraction of manures derived from animal houses. Pathogen reduction may be enhanced by digestion in biogas reactors or composting. The solid manure is often composted in Asia to reduce pathogens, parasites, insects and the burden of weed seeds; also, to improve handling of the solid manure. However, composting causes a significant reduction in the N content of the manure, in particular, of the more readily available inorganic N fraction.

It has been observed that additives (*e.g.* yucca extract to reduce NH_3 emission and ‘efficient microorganisms’ to reduce odour) are highly valued in Asia for improving the fertilizer value of manure and reducing the emissions of malodorous gases. However, there is little scientific documentation available concerning the effect of these additives.

Not much is known about the nature of manure processing techniques in current use and the extent of manure additives in use, but it appears that the storage and processing techniques are only adopted if there is no immediate demand for the manure as fertilizer for the existing crops. Otherwise, the manures tend to be spread directly, without further storage or pre-treatment. Furthermore, there are few statistics available relating to manure handling systems. Because of the limited information on manure handling and utilization systems in Asian countries, general concepts on sustainable manure management are presented in this Manual.

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3. A SYSTEMS APPROACH TO SUSTAINABLE MANURE MANAGEMENT

Livestock production has expanded rapidly in many Asian countries and this expansion is expected to continue in the years to come (Chapter 2). The growth of meat and milk production is demand-driven, and this is fuelled by increasing incomes, urbanization and changing lifestyles. The expansion of livestock production increases the demand for land to produce high-quality feeds and forages as well as the production of animal manures. Questions arise about the number of animals that can be kept in a country or in a region without detrimental effects on food security, natural resources and environmental quality, and human and animal health. To answer these questions, several aspects have to be considered, *viz*. production and supply of feeds and forages, manure management and utilization, and control of the content of undesirable substances in the manure: heavy metals, antibiotics, veterinary medical residues, parasites and pathogens, and weed seeds. Important criteria for the development of sustainable livestock production will be discussed in this chapter. Developments of the livestock sector in The Netherlands in the second half of the 20th century will be described as an illustration of the Livestock Revolution in the Western world and the measures taken to limit the negative effects on the environment.

Environmentally friendly management of livestock manures should be an integral part of efforts to improve the sustainability of agriculture. Experiences in Europe and the USA have shown that land application of livestock manures for the fertilization of crops and grasslands and for improvement or maintenance of soil fertility is the most suitable method of manure utilization [1]. Up to now, industrial or on-farm systems of manure processing have hardly developed because of the high costs. We do not see reasons why this should be different in Asian countries.

3.1. DEVELOPMENTS OF CROP YIELDS AND THE EFFICIENCY OF THE PRODUCTION PROCESS

An interesting analysis of the increase of crop yields in the 20th century and its significance for food supply, efficiency of the production process, and environmental quality, has been presented by De Wit and co-workers [2–4]. These authors focused attention to the fact that a few years after the Second World War, yield increase of arable crops in the Western world showed a sudden change. For instance, between 1900 and 1950 yields of winter wheat in the USA and the UK increased by 3 and 4 kg ha^{-1} year^{-1}, respectively, and between 1950 and 1980 by 50 and 78 kg ha^{-1} year^{-1}. A similar change occurred approximately 20 years later in some developing countries in Asia, *e.g.* in Indonesia, where the annual yield increases of paddy rice suddenly changed from 2.5 kg ha^{-1} to *ca.* 130 kg ha^{-1}. This change from a low rate of increase of crop yields to a high rate, the so-called ‘Green Revolution’, was caused by an innovative combination of plant breeding, plant nutrition, water management, control of weeds, pests and diseases, and mechanization [2]. The introduction of new technologies was stimulated by the agricultural policy of governments and international organizations. The mentioned rates of increase of the yields of cereals in traditional agriculture were less than 0.3% per year and generally much lower than the rates of population growth. As a consequence, an expansion of the cropped area was necessary for the supply of sufficient food. The rate of increase of crop yields after the Green Revolution was considerably higher than the rate of population growth, particularly in the first decades when average yield levels were still low. This decreased pressure on the land, and allowed the use of land for the production of vegetables, fruits, non-food crops and high-quality feeds and forages for livestock. In addition, some marginal lands could be reforested.

The increase of crop yields caused by the Green Revolution, and the growing demand for livestock products were essential conditions for the development of the so-called Livestock Revolution (Chapter 2). The Livestock Revolution is characterized by an increase of livestock numbers and production per animal. The latter was caused by the simultaneous improvement of animal breeding, animal nutrition, animal health, and housing and mechanization. As a consequence, the rate of increase of milk production per cow in the USA and The Netherlands averaged *ca.* 100 kg $year^{-1}$ after 1970 [5]. However, the rate of increase in New Zealand was much lower, *viz.* only *ca.* 35 kg cow^{-1} $year^{-1}$, probably because of the limited use of concentrates in this grassland-based production system. Apparently, ample availability of good concentrate ingredients at low costs is very important for the development of productive animal production systems.

Intensification of crop and animal production is not necessarily harmful for the environment. De Wit and co-workers showed that in crop production the simultaneous improvement of different production factors is very effective in terms of resource use efficiency [2–4, 6]. In many cases there is a positive interaction between inputs. For instance, good drainage of the soil is important for the performance of many crops, but has also positive effects on the possibilities for timely mechanization and on the efficiency of N utilization [7]. It is common knowledge that an increase in crop yield decreases the fixed costs per unit of product. De Wit [3] adds to this that *'yield increases due to technical advance may often require more of some inputs per unit area, but, at the same time, require less of most inputs per unit product. Innovations that lead to yield increases are therefore advantageous under most economic regimes, provided that the crop can be grown economically. Hence, where a crop can be grown economically, the yield per hectare will continue to increase until the level that climate, soil, reclamation level, and know-how permit'*. De Wit [3] concludes that in high yielding situations, less external inputs are required and wasted for a certain total production.

The preceding theory should be examined carefully. Although several examples are given to show its general validity [4], it seems in flat contradiction with the strong increase of environmental problems associated with the intensification of crop and animal production. Apparently, this is caused by 2 characteristics of modern agriculture, mentioned below:

(1) The liberal use of relatively cheap external inputs, like chemical fertilizers. In modern agriculture, chemical fertilizers are cheap compared to other means of production [8, 9]. As a consequence, the economically optimum rates of application are high and often some extra fertilizer is given to be sure that deficiencies do not occur. According to the Law of Diminishing Returns, the efficiency of nutrient use decreases and emissions increase strongly when the rate of application approaches or exceeds the economic optimum [5].

(2) The industrialization and specialization of agricultural production which often caused separation of crop and animal production. In traditional agriculture, animal manures had a very important role in crop nutrition and maintenance of soil fertility, and crop yields depended largely on plant nutrients provided by animal manures, soil reserves and biological N fixation. Under these conditions, effective utilization of animal manures was essential for food production. Since the 1960s, ample availability of chemical fertilizers at low costs has decreased the interest to utilize livestock manures as a source of nutrients for crops. Moreover, storage, handling and application of chemical fertilizers are easier and less costly, and the availability of nutrients, in particular of N, is often more reliable (Chapter 7). As a consequence, animal manures have been replaced by chemical fertilizers in crop production, and are being wasted causing pollution of atmosphere, soil and water.

Similarly, crop and food residues have been replaced by industrial concentrates in livestock production.

It is evident that the mentioned observations of De Wit and co-workers apply to situations of judicious management and fertilization of individual crops. However, this is a too limited frame of reference because agriculture generally includes both crop and livestock production and nutrient use efficiency and environmental impact should be assessed in the whole soil-crop-livestock system. Moreover, agricultural practice may differ from theory due to the fact that efficient utilization of plant nutrients is often not an important objective of farmers.

Livestock production is based on crop production, *i.e.* on forages, grains and by-products of the crop processing industry. This means that all animal production systems require land for feed production. That land may be on the livestock farm itself and we call those animal production systems 'land-based'. Feeds may also be produced on other farms, even in other countries, and we call those livestock farms 'land-less'. Many specialized animal production farms are partly land-based: they grow a part of the feed requirements on the farm and buy the remaining part from outside. Livestock production also requires land and crops for a proper (sustainable) utilization of animal manures. On land-based livestock farms, animal manure generally can be used for forage and feed production. Landless animal production units should find other farms without animals or with a low animal density which are prepared to include animal manure in the fertilization plan of the crops. This is essential for sustainable utilization of manures produced on land-less livestock farms.

Similarly, crop residues and by-products of the food industry, in particular those with a high content of plant nutrients, should be utilized as much as possible in livestock rations.

3.2. INTENSIFICATION OF LIVESTOCK PRODUCTION IN THE NETHERLANDS

In the period between approximately 1950 and 1985, national policies and the Common Agricultural Policy of the European Community strongly stimulated production and domestic self-sufficiency of food in Western Europe [2]. This was triggered by food shortages during and shortly after the Second World War and stimulated by the increase in prosperity and the demand for more luxurious food in the following decades. The essence of agricultural policy in this period was that farming should be supported by the state, and this in turn would ensure the well being of rural areas [10]. Especially in The Netherlands, the efforts to increase agricultural production have been very successful. In particular, the following measures contributed to this:

- Effective research, advisory and education programs.
- Improvement of rural infrastructure.
- Reclamation and improvement of land, with measures such as re-allotment, land drainage and leveling, and improvement of farm infrastructure.
- Promotion of ample supplies of production inputs, including credits and, in some cases, subsidies on investments.
- Guaranteed product prices.

These supporting measures, as well as the favorable climate and topography, and the strategic situation of Dutch agriculture regarding markets and supplies, particularly stimulated the growth of animal husbandry (dairy, pig and poultry production) and horticulture (vegetables, flowers, ornamental plants). The following developments strongly contributed to the expansion of livestock numbers and production:

- *Increased use of imported concentrates.* After approximately 1960, livestock numbers and production per head could increase as a result of the ample supply of (mostly) imported concentrate ingredients at relatively low costs. This was favored by the proximity of the harbor of Rotterdam and the good transport infrastructure of the hinterland. As a consequence, pig and poultry production developed mainly on imported feeds in almost landless units. Dairy farming remained largely land-based, although imported concentrates increased milk production per cow and allowed higher animal densities on many farms than the available area of land could support. In general, forages were produced on-farm, whereas concentrate ingredients were produced elsewhere. Total consumption of manufactured concentrates by Dutch livestock increased to *ca.* 18 million metric tons per year in the 1980s. Between 75 and 80% of the concentrate ingredients were imported from other countries, containing *ca.* 420 million kg N, 70 million kg P and 165 million kg K [11, 12].
- *Increased use of artificial fertilizers*, in particular of N on grasslands. Extensive research programs were carried out between 1960 and 1980 to determine the effects of fertilizer N on herbage yield and quality [8, 13]. Prins [13] concluded from the results of cutting experiments that, at an assumed marginal profitability of 7.5 kg DM per kg N applied, the optimum rate of N for grassland on sand and clay soils in The Netherlands was 420 (range, 360–520) kg ha^{-1} $year^{-1}$. This was close to the official recommendation at that time of 400 kg N ha^{-1} $year^{-1}$ (effective N from animal manure + fertilizer N) on sand, clay and wet peat soils, and 250 kg N ha^{-1} $year^{-1}$ on drained peat soils. The average use of fertilizer N on grassland increased from *ca* 50 kg ha^{-1} in 1950 to as much as 315 kg ha^{-1} in 1985. The development of the use of chemical fertilizers in Dutch agriculture is shown in Table 3.1. This Table shows a strong increase of the use of nitrogenous fertilizers between 1950 and 1980 and a decrease afterwards, illustrating the growing awareness of environmental pollution and related legislation. Consumption of P and K containing chemical fertilizers decreased since 1950, probably as a result of the increased availability of animal manures and the practice of regular soil sampling and analysis on many farms.
- *Mechanization and automation* of the whole production process to replace expensive labour and improve management. Important developments were: (1) modern livestock housing systems with facilities for collection, storage and application of liquid manure (slurry), (2) manufacturing and handling of concentrates, (3) conservation of herbage and fodder crops as silage, and (4) milking machines and, recently, milking robots.

TABLE 3.1. DEVELOPMENT OF THE USE OF CHEMICAL FERTILIZERS IN DUTCH AGRICULTURE (IN MILLION KG ELEMENT PER YEAR). SOURCE [14]

Element	1950	1980	1990	2000	2005[1]
N	156	485	412	339	279
P	52	36	33	27	21
K	128	93	81	70	n.a.[2]

[1] Provisional data; [2] Not available

Growth of livestock production in The Netherlands is illustrated in Table 3.2. Almost all the sectors showed a strong increase between 1950 and 1990. Milk and beef production decreased slightly in the last decades, owing to the introduction of the milk quota system by the European Community in 1984 [5].

TABLE 3.2. DEVELOPMENT OF THE PRODUCTION OF MILK, MEAT AND EGGS IN THE NETHERLANDS. THE VALUES ARE THREE-YEAR MOVING AVERAGES, CENTERED ON THE YEARS SHOWN. SOURCE: [14]

Year	Production (in million kg) of				
	Milk	Beef	Pig meat	Poultry meat	Eggs
1910	2,639	n.a.[1]	n.a.[1]	~	n.a.[1]
1930	4,480	ca. 150	ca. 270	~	125
1950	5,638	133	222	~	113
1960	6,657	241	403	67[2]	286
1970	8,189	332	741	275	248
1980	11,867	444	1,350	382	542
1990	11,244	397	1,857	521	659
2000	10,780	n.a.[1]	1,807	664	628

[1] Not available; [2] Broiler production started in the 1950s

In 1950, the livestock population in The Netherlands consisted of 2.73 million head of cattle (including 1.52 million lactating dairy cows) and 1.86 million pigs. Poultry numbers were not yet recorded in that year, but amounted to 38 million laying hens and 4.5 million broilers in 1960. In 1990, these numbers were 4.93 million head of cattle (including 1.88 million lactating dairy cows), 13.9 million pigs, 44 million laying hens, and 41 million broilers [14]. The numbers presented indicate the standing population at the moment of the agricultural census (May), often referred to as animal places. The number of dairy cows decreased by *ca*. 40% since 1984 due to the limitation of total milk production by the milk quota system and the increasing production per cow. The number of pigs decreased after 1997 because of outbreaks of classical swine fever in 1997/1998 and foot and mouth disease in 2001 [15]. The numbers of laying hens and broilers decreased after 2002 by an outbreak of avian influenza in 2003. Since 1990, the expansion of pig and poultry production was increasingly limited by environmental legislation [16].

The total area of agricultural land in 1990 amounted to approximately 2 million hectares [14]. This included *ca*. 1.1 million ha of grassland, 0.2 million ha of silage maize, 0.6 million ha of arable land (mainly potatoes, cereals, sugar beets and field vegetables), and 0.1 million ha of horticulture (vegetables, flowers, ornamental plants and fruits).

3.3. CONCERN ABOUT MANURE SURPLUSES

A consequence of the expansion and intensification of the Dutch livestock production, that received much attention since the late 1970s, was the excessive production of animal manures in some regions. This problem was first quantified in a case-study of the situation in the region De Peel in the province of North Brabant [17]. De Peel comprises 85,000 ha, 50,000 ha of which is agricultural, *viz*. about 30,000 ha of grassland, 14,000 ha of silage maize and 6,000 ha of arable and horticultural crops. The area has a very high livestock density with a calculated average production of manure nutrients in 1984 of 714 kg N, 133 kg P and 558 kg K per ha of agricultural land. This is far in excess of the requirements of local crops. A large fraction of the manure produced was applied as slurry to the maize land on the basis of the experience that excessive slurry applications do not damage this crop but, on the contrary, improve its yield. However, excessive rates of animal slurry on maize land cause large nitrate leaching losses and accumulation of P and heavy metals in the soil and, on the long term, leaching of P and heavy metals and, possibly, too high concentrations of some heavy metals in the products [17, 18]. This case-study showed that De Peel had a serious manure surplus and that transport of manures to areas with a much lower livestock density (and a 'manure shortage') was necessary. Manure transport from De Peel to other areas was organized by the North-Brabant 'manure board' ('manure bank'), a provincial organization set up for this purpose [17]. While there was some subsidy on manure transport in the first years, farmers with a manure surplus had to pay an increasing part of the transport costs and, in situations of limited demand for manure, a fee to the farmer who accepts the manure. As a consequence, proper disposal of manure became rather expensive for landless livestock farms. This stimulated the development and introduction of technologies to minimize manure production, both in terms of volume and nutrients (Chapter 4).

Calculations of manure production at national level in The Netherlands started approximately in 1990. Excretion of N by different livestock sectors in 1989 is presented in Table 3.3, and excretion of P in Table 3.4 [11, 12, 19]. These calculations are based on national statistics on the use and composition of forages and concentrates, livestock numbers and production, and N and P contents of products. It is also possible to calculate production of manure N and P on the basis of livestock numbers and standardized values for the N and P excretion per animal (Chapter 4).

TABLE 3.3. AMOUNTS OF N IN FEEDS CONSUMED, PRODUCTS AND EXCRETA OF THE MAIN LIVESTOCK SECTORS IN THE NETHERLANDS IN 1989. DATA ARE IN MILLION KG OF N

Livestock sector	N inputs		N in products	N excreted	N efficiency (%)[2]
	Local feeds[1]	Imported			
Cattle	402	124	81	445	15
Pigs	30	165	57	138	29
Poultry	18	82	31	69	31
Total	450	371	169	652	

[1] Forages, byproducts of the primary production sectors and agro-industry, synthetic amino acids; [2] N efficiency = N in products/N inputs

Total N excretion in faeces and urine amounted to 652 million kg per year (Table 3.3); this is equivalent to 1,450 million kg of urea (45% N). Not all this N is available for field application to crops and grasslands, due to gaseous N losses from livestock buildings and stored manure [20]. Another part of excreted N is voided by grazing cattle in urine on the grasslands and, because of the poor distribution and large losses, this N is generally not taken into account in the fertilizer planning and recommendations. Total P excretion amounted to almost 110 million kg (Table 3.4); this is equivalent to approximately 600 million kg of triplesuperphosphate (42% P_2O_5). Excreted P is less susceptible to losses than excreted N and, unless it is applied in excessive amounts, it can be fully utilized by the crops.

TABLE 3.4. AMOUNTS OF P IN FEEDS, PRODUCTS AND EXCRETA OF THE MAIN LIVESTOCK SECTORS IN THE NETHERLANDS IN 1989. DATA ARE IN MILLION KG P

Livestock sectors	P inputs		P in products	P excreted	P efficiency (%)[2]
	Local feeds[1]	Imported			
Cattle	54.0	22.2	16.2	60.0	21
Pigs	14.4	28.4	11.9	30.9	28
Poultry	8.2	15.5	4.7	19.0	20
Total	76.6	66.1	32.8	109.9	

[1] Forages, byproducts of the primary production sectors and agro-industry, synthetic P supplements; [2] P efficiency = P in products/P inputs

Cattle had the largest share in total N and P excretion *viz*. 68 and 55%, respectively (Tables 3.3 and 3.4). However, cattle farms occupied *ca.* 1.3 million ha of land in 1989, whereas pig and poultry farms only had *ca.* 50,000 ha [21]. As a consequence, many pig and poultry farms had problems to dispose of manure, even before the introduction of environmental legislation (Section 3.6). This was aggravated by the fact that a large part of pig and poultry production was concentrated in areas with intensive dairy farming, *viz*. on the sandy soils in the Eastern and Southern provinces of the country [17].

A good appreciation of the amount of livestock manure and manure nutrients can only be made in comparison with the requirements or removal of nutrients by the crops, and the use of chemical fertilizers. Table 3.5 presents a comparison of the N and P supplies in livestock manures and chemical fertilizers to the *ca*. 2 million hectares of agricultural land in The Netherlands, and the N and P removals in harvested crops. The supplies of manure N and P are taken from the Tables 3.3 and 3.4. Data on the use of chemical fertilizers and N and P removals in crops have been taken from agricultural statistics [11, 12, 19].

The N and P surpluses, presented in Table 3.5, are estimates of the losses of N and P from the agricultural land to the environment + changes in the N and P contents of the soils. Most of the N surplus will be lost to the environment by gaseous losses and nitrate leaching. Most of the P surplus will initially accumulate in the soils, but with increasing saturation of the phosphate adsorption capacity of the soils, an increasing part will be leached to ground and surface waters [22].

TABLE 3.5. NITROGEN AND P BALANCES OF AGRICULTURAL LAND IN THE NETHERLANDS IN 1989. SOURCE: [19]

	N (million kg per year)	P (million kg per year)
Inputs:		
Animal manures	489 [1]	110
Chemical fertilizers	444	38
Total	933	148
Outputs:		
Crops (net)[2]	470	67
Surplus (inputs-outputs)	463	81

[1] N excretion minus 25% to account for gaseous N losses from livestock buildings, stored manure and dung and urine of grazing animals (Chapter 5); [2] N and P removed in harvested products

The data presented in Table 3.5 question the use of large amounts of chemical fertilizers in Dutch agriculture. Better distribution of animal manures over the 2 million hectares of agricultural land in combination with better techniques and timing of manure application will allow considerable reductions in the use of chemical fertilizers and consequently in N and P losses. In addition, better animal feeding practices may reduce the production of manure N and P and better crop management may increase N and P removal in harvested crops.

Production of manure nutrients should be in equilibrium with the capacity of the crops to utilize them, *i.e.* with nutrient removal in the harvested crops. This can be considered as the ecological livestock carrying capacity of agricultural land. Production of manure P generally limits the livestock carrying capacity. This is illustrated in Table 3.5, showing a relatively greater surplus of manure P than of manure N compared to the P and N removals in the harvested crops. In other words, the mean N/P ratio in livestock manures (4.45) is smaller than the mean N/P ratio in crops (7.01). Table 3.5 shows that the supply of manure P strongly exceeded the removal of P in harvested crops, indicating that the livestock population in The Netherlands was not in equilibrium with the capacity of the crops to utilize manure P and that P accumulated in the soils. For some time, this can be positive because it improves the P status of the soils, but on the longer term this causes P leaching and eutrophication of surface waters.

3.4. NUTRIENT BALANCES AS A TOOL TO ASSESS ENVIRONMENTAL PROBLEMS

In the late 1970s, public concern for the negative effects of intensive animal production systems on the environment concentrated on the so-called ‘manure surpluses’ in areas with mainly landless pig and poultry production (Section 3.3). However, studies in the early 1980s also revealed potentially large N losses from land-based dairy farms. So-called farm-gate N balances were proposed to assess the efficiency of N use on these farms and potential N losses to the environment [23]. A farm-gate N balance is a balance sheet of all inputs of N on the farm (‘through the gate’) and all outputs of N in products exported. On a dairy farm, N inputs take place *via* chemical fertilizers, livestock manures (as far as they are ‘imported’ from other farms), purchased feedstuffs (roughages and concentrates), atmospheric deposition and biological N fixation. Nitrogen outputs take place *via* the ‘export’ of animals, animal products, crops and manure from the farm. The difference between N inputs and N outputs is the N surplus. The N surplus of a farm-gate balance is an estimate of the losses of N to the environment + the change in N ‘stocks’ on the farm. The main N ‘stock’ on a farm is in soil organic matter, but other stocks are in stored feedstuffs, *e.g.* silage, hay and concentrates, and stored manure, which may slightly change from year to year. Nitrogen inputs, outputs and surplus are generally expressed in kg N ha^{-1} $year^{-1}$ to allow between-farm comparisons.

The concept of the farm-gate N balance was first applied in The Netherlands to compare the N use efficiency of grassland-based dairy farms with intensive and extensive management (Table 3.6; [23]). This was undertaken to answer the question whether the intensively managed farms were causing more environmental pollution than the extensively managed farms (Section 3.1). The intensive farms assessed, the Nitrogen Pilot Farms, were managed in accordance with the recommendations of those years. They particularly were keen to follow the official recommendations for N, P and K fertilization of grassland, grassland management and animal feeding [8]. The extensive farm was a well-managed bio-dynamic farm, relying almost exclusively on the farm-yard manure produced and white clover for nutrient supply to the grassland, and on home-grown forages for animal feeding (Table 3.6).

TABLE 3.6. FARM-GATE N BALANCES OF INTENSIVELY AND EXTENSIVELY MANAGED GRASSLAND-BASED DAIRY FARMS IN THE NETHERLANDS IN 1975/1976 (VALUES ARE KG N HA^{-1} $YEAR^{-1}$). SOURCE: [23]

	Intensive	Extensive
N inputs:		
- chemical fertilizers	383	-
- biological fixation	-	65
- purchased feeds	127	24
- atmospheric deposition	23	23
Total	533	112
N outputs:		
- milk	72	31
- sold animals	12	7
Total	84	38
N surplus (inputs – outputs):	449	74

Annual production on the extensively managed farm amounted to 5,870 kg milk and 250 kg liveweight gain per ha, whereas it averaged 13,500 kg milk and 470 kg liveweight gain per ha on the Nitrogen Pilot Farms [23]. Hence, the 2.2-fold increase in animal production on the Nitrogen Pilot Farms (in terms of N output, Table 3.6) was accompanied by a 6-fold increase in N surplus. From the data presented, it can be calculated that only 11% of the extra N input on the Nitrogen Pilot Farms compared to the bio-dynamic farm was recovered in animal products. The following points have been identified as the main causes of this low N efficiency [23, 24]:

- The negative correlation between the rate of N application and biological N fixation. Applied N stimulates grass growth and favours grass in the competition with white clover. This reduces clover growth and related N fixation [25–27]. The high rates of N application on the Nitrogen Pilot Farms practically eliminated white clover from the swards and, consequently, biological N fixation (Table 3.6).
- The poor utilization of manure N on the intensively managed farms. Due to the ample availability of easy-to-handle nitrogenous fertilizers at low costs, animal manure was disposed of as cheaply as possible. Generally, very high rates of slurry were applied in autumn and winter to a limited number of fields, in particular to the fields intended for grassland renovation and production of silage maize in the next growing season [18, 28].
- Increasing rates of N have a decreasing effect on herbage yield ('law of diminishing returns') but increase herbage N content. Unless this herbage is supplemented with low-protein/high-energy forages or concentrates, this causes excessive N contents in the diets of the animals and increased excretion of N in faeces and, particularly, in urine. At the same time, increased N application rates reduce the re-utilization of excreted N because of the reduced capacity of the sward and microbial biomass in the soil to act as sinks.
- Almost all the research on the effects of applied N on herbage yield was carried out in small-plot cutting experiments and little attention was paid to the response of grassland and animal production to applied N under farming conditions. Possibly, on-farm effects of applied N were smaller than expected.

The N balances of the Nitrogen Pilot Farms revealed the low N use efficiency and potentially high N losses of dairy farms under the management in those years (Table 3.6).

This marked the start of extensive research to quantify N flows and losses on these farms and to identify possibilities to improve N use efficiency. The most important aspects of this were:

- Better utilization of manure N in grass and forage crop production and corresponding reduction in the use of fertilizer N [28, 29].
- Better utilization of applied N by the development of site-specific recommendations for N application to grassland and forage crops, taking into account local growing conditions and N supply from the soil [7, 30].
- Better utilization of dietary N by the animals by the formulation of balanced diets without N surpluses [31, 32].

In The Netherlands, farm-gate N balances have become important tools to assess the efficiency of N utilization of farms and potential N losses to the environment. They have been used in research, farm advisory work and farm management, as well as in environmental legislation [16]. Similarly, farm-gate P balances have been used to assess the efficiency of P utilization. Since approximately 1985, farm-gate N and P balances have been drawn up of the dairy farms included in the Dutch Farm Accountancy Data Network (FADN) of the Agricultural Economics Research Institute [33]. The objective of this was to monitor the developments in nutrient management on these farms. Table 3.7 has been derived from FADN and presents the average N, P, and K balances of the 175 specialized dairy farms monitored on sandy soils [34]. These balances are averages over the period 1983–1986, when inputs of nutrients as well as milk production per ha were at their highest level.

TABLE 3.7. AVERAGE ANNUAL NUTRIENT BALANCES IN 1983–1986 OF 175 SPECIALIZED DAIRY FARMS ON SANDY SOILS IN THE NETHERLANDS. SOURCE [34]

	Element (kg ha^{-1} $year^{-1}$)		
	N	P	K
Inputs:			
- chemical fertilizers	331	15	20
- purchased concentrates	137	25	74
- purchased roughage	44	6	34
- atmospheric deposition	48	1	4
- miscellaneous	8	1	4
Total	568	48	146
Outputs:			
- milk[1]	67	12	19
- sold livestock[2]	14	4	1
- sold roughage	1	0	0
Total	82	16	20
Surplus (Inputs-Outputs)	486	32	126

[1] About 13,000 kg ha^{-1}; [2] About 540 kg ha^{-1}

Table 3.7 shows that chemical fertilizers contributed most to the surplus on the N balance, and purchased feeds to the surpluses of P and K. The N surpluses of these farms reflect large N losses by ammonia volatilization, denitrification and nitrate leaching. The P surpluses indicate P accumulation in the soils and, on the long term (depending on the cumulated P surplus and the P adsorption capacity of the soil), leaching of phosphates. A small part of the K surpluses may accumulate in the soil, but because of the small cation exchange capacity of the sandy soils on these farms, the major part will be lost by leaching. Up to now, K accumulation on dairy farms only received attention because of related animal health problems, in particular grass tetany or hypomagnesemia [8, 35]. The N and P surpluses in Dutch agriculture in the 1980s (Tables 3.5 and 3.7) forced the Dutch government to enact environmental legislation (Section 3.6).

Farm-gate nutrient balances are particularly useful for land-based livestock farms, *i.e.* for farms with both crop and livestock production. Examples of such production systems are the land-based dairy farms in The Netherlands (Tables 3.6 and 3.7), and arable farms with a pig or poultry production unit, which is a common farm type in Denmark. The nutrient surpluses of this type of farms are good indicators of the efficiency of nutrient use in the whole soil-crop-livestock system. Effective utilization of manure nutrients on these farms may reduce the N and P inputs *via* chemical fertilizers as well as the N and P surpluses. Similarly, effective utilization of the home-grown forages, feeds, and crop residues allows a reduction in the input of N and P *via* purchased forages and concentrates.

Although nutrient balances are also drawn up for land-less livestock farms and arable farms, they have limited value for assessment of the environmental impact of these farms. The figures on the N and P balances of landless livestock farms are generally expressed in kg per year for the whole farm. These balances include the inputs of N and P *via* purchased feeds, animals, and bedding material and the outputs in sold animals and exported manure. The N and P surpluses equal the sum of N and P losses from the buildings, the farm-yard and the stored manure. However, these balances do not account for the efficiency of re-utilization of manure nutrients for crop production.

The nutrient balance of an arable farm quantifies inputs *via* atmospheric deposition, livestock manures, chemical fertilizers, biological N fixation, and seeds/planting material and outputs *via* products, by-products and crop residues. However, these balances do not indicate whether local N sources, in particular livestock manures and composts, are properly utilized. Arable production systems may have a deficit in the balance of one or more elements. This is often the case in regions with a low livestock density, and indicates depletion of soil fertility and potential soil degradation.

3.5. OBJECTIVES FOR ENVIRONMENTAL QUALITY

Measures to mitigate nutrient losses from livestock farms should be based on clear public objectives for environmental quality. Formulation of these objectives is difficult because it requires a political process of weighing and compromising conflicting objectives. On the one hand, there is an urgent need in many countries to increase livestock production and to improve income of livestock farmers. On the other hand, it is extremely important to protect the natural resources soil, water, atmosphere and biodiversity, and to ensure a good environmental quality, healthy natural ecosystems and attractive landscapes. Protection of soils is necessary for food security of future generations. Water should be protected to serve as drinking water, to be used for recreation, and to sustain aquatic production systems and wildlife. Pollution of the atmosphere should be reduced because it threatens human and animal health and contributes to climate change. Biodiversity has an economic aspect (genes for the future) as well as an ecological function (health of ecosystems). Developed nations consider environmental quality as a very important aspect of the quality of life [36]. These general ideas should be taken into account in an early stage of intensification of crop and livestock production and should be important considerations in the development of environmental policy.

Water quality has received much attention in environmental protection in Europe. Pollution of drinking water sources by nitrates is of serious concern in most Member States of the European Union. According to international standards, nitrate concentration in ground and surface water that can be used for the preparation of drinking water should not exceed 50 mg per litre, *i.e.* 11.3 mg nitrate-N per litre; Table 3.8 [37, 38]. In addition, a target concentration of 25 mg nitrate per litre has been established. If the nitrate content exceeds 50 mg per litre, nitrates have to be removed, which is a very costly process [5]. The critical nitrate concentration of 50 mg per litre is a difficult target for farms in areas with a small or moderate precipitation surplus and freely drained soils with a limited denitrification capacity. For instance, in regions with a precipitation surplus of 300 mm $year^{-1}$, like in The Netherlands, nitrate leaching losses should not exceed 34 kg N ha^{-1} $year^{-1}$.

In The Netherlands, the critical value of 50 mg nitrate per litre applies to all groundwater resources that potentially can be used for drinking water, *i.e.* water with less than 150 mg Cl^- per litre. Moreover, critical values have been defined for average N and P contents in stagnant surface waters in summer (Table 3.8). These are 2.2 and 0.15 mg per litre, respectively. Finally, international agreements on a reduction in total N and P emissions to the North Sea have to be observed. For both elements, this reduction amounts to 50% of the 1985 level. In the near future, national policies for surface water quality will be increasingly affected by the European Water Framework Directive [39]. This requires a 'good ecological condition' of all water resources of the Member States in 2015.

TABLE 3.8. OBJECTIVES FOR WATER QUALITY IN THE NETHERLANDS (MG PER LITRE) [40]

Parameter	Groundwater		Surface water	
	Maximum	Target value	Maximum	Target value
Total N	-	-	2.2	1
Total P	-	0.4/3[1]	0.15	0.05
Nitrate	50	25	-	-
Ammonium-N	-	2/10[1]	-	-

[1] Lowest value for sand, highest for clay and peat

Ammonia volatilization contributes strongly to the high rates of atmospheric N deposition in The Netherlands and other West European countries [41, 42]. On average, N deposition in The Netherlands amounted to 38 kg ha^{-1} in 1993, of which about 72% was as ammonia or ammonium salts, together indicated as NH_x [43]. In some areas with a high livestock density, average NH_x deposition was as high as 70 kg N ha^{-1}. Similar values have been reported in the United Kingdom [44]. After volatilization, about 30% of the ammonia returns as wet or dry deposition to soils and vegetations within 5 km of the source. A large part of the remaining 70% reacts in the atmosphere with SO_2 and NO_x and is transported over a distance of 5 to about 1000 km [43]. High rates of N deposition cause ecological damage to forests and nutrient-poor natural ecosystems [45, 46]. These vegetations absorb and accumulate this N effectively [47]. The resulting increase of N supply causes undesirable floristic changes, loss of biodiversity and physiological problems to trees, such as increased susceptibility to abiotic and biotic stress (drought, frost, herbivory, fungal diseases) and deficiencies of other nutrients. Besides, deposition of NH_x potentially contributes to soil acidification which may also affect vegetation. This acidifying effect only occurs after nitrification of NH_x in the soil, particularly when part of the nitrates produced is lost by leaching [43, 48].

Based on assessments of ecologically acceptable values for acid and N deposition in different natural ecosystems, the Dutch government aims at a reduction in the average acid deposition from 4280 mol H^+ ha^{-1} in 1993 to 1400 mol H^+ ha^{-1} in 2010. Simultaneously, average atmospheric N deposition should be reduced from 38 kg ha^{-1} in 1993 to 14 kg ha^{-1} in 2010 [49]. Related to this, ammonia volatilization from the animal production sector has to be reduced by 50–70%, compared to 1980. The national policy is enforced by the EC Directive on National Emission Ceilings for certain Atmospheric Pollutants [50] and the United Nations Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (Gothenburg Protocol; [51]). Further reductions in ammonia emission in The Netherlands are envisaged for 2030 [36].

In the atmosphere, ammonia reacts with SO_2 and NO_x. The reaction products, ammonium sulphate and ammonium nitrate, are two prevalent forms of secondary particulate matter in the atmosphere [52, 53]. In Europe, secondary particles constitute 50% or more of $PM_{2.5}$, and 50 to 90% of PM_{10}. PM_{10} has been indicated as 'thoracic' particles of $\leq$ 10 micrometer (μm), that can penetrate into the lower respiratory system; $PM_{2.5}$ as 'respirable' particles of $\leq$ 2.5 μm, that can penetrate into the gas-exchange region of the lung [54]. There is a considerable body of evidence that exposure to fine particulate matter in the air is associated with increases in mortality and hospital admissions due to respiratory and cardiovascular disease [54, 55]. In Europe, the policy to reduce exposure of humans to particulate matter aims at a reduction in the concentration of PM_{10}. For 2010, the targeted maximum values for PM_{10} in the air are 20 μg m^{-3} as an annual average and a maximum of seven daily exceedances of 50 μg m^{-3} [53]. Reductions in precursor emissions (NH_3, NO_x and SO_2), in particular of NH_3, will contribute significantly to reductions in secondary PM concentrations [53].

Intensive livestock farming generally contributes to the accumulation of heavy metals in soils. Some heavy metals, in particular copper (Cu), and zinc (Zn) are essential minerals for farm animals. Although Cu and Zn requirements of most livestock categories can be completely or almost completely met by the feed ingredients, it is common practice to add additional Cu and Zn *via* mineral mixtures. This generally results in a large oversupply. Reasons for this oversupply are [56]: (1) the positive effect of Cu on pig performance, in particular on farms with 'sub-optimal' management; (2) the problems to establish precise minimum requirements because of interactions with other minerals. Other heavy metals, like cadmium (Cd), chromium (Cr), mercury (Hg), lead (Pb) and nickel (Ni) are nothing but pollutants. Livestock farms import heavy metals *via* purchased feeds, chemical fertilizers, sewage sludge and other types of waste. Large fractions (generally > 90%) of the heavy metals in livestock diets are excreted in manure. Consequently, the concentrations of heavy metals in manure strongly depend on the concentrations in the feeds consumed. Most heavy metals accumulate in agricultural soils. A recent study of the soil-crop balances of the heavy metals Cd, Cr, Cu, Hg, Ni, Pb, and Zn in agricultural soils in The Netherlands showed surpluses (*i.e.* inputs *via* livestock manures, chemical fertilizers, composts, *etc*. minus outputs in harvested crops) of all the metals studied [57]. Inputs were between 2 and 5 times higher than outputs. This is a point of concern, because accumulation of heavy metals increases their availability and uptake by plants as well as leaching to groundwater and surface water [22]. This may have negative effects on food quality and human health as well as on the health of aquatic ecosystems. A prerequisite to sustainable agriculture is to control inputs of heavy metals in such a way that soil and water functions and product quality will not be impeded in the future [58, 59]. The Commission of the European Communities has established maximum values for the intake of iron (Fe), cobalt (Co), copper (Cu), manganese (Mn) and zinc (Zn) by different livestock categories [60]. These values are based on the physiological requirements of the animals and aim to restrict oversupply. Other toxic substances that require attention in livestock production are antibiotics, hormones, and veterinary medical residues.

Livestock farming causes considerable emissions of the greenhouse gases CO_2, CH_4 and N_2O. These emissions received much attention in recent years because of the alleged contribution of these gases to global climate change. Livestock production causes direct emissions of greenhouse gases, *e.g.* of CO_2 by combustion of fossil fuels on the farms, but also by digestion or decomposition of organic matter in animals, stored manure and soils. In addition, CH_4 is emitted by animals, and as the end-product of anaerobic decomposition of organic matter. Nitrous oxide is produced during nitrification or denitrification in stored manure and soils. However, there is also a need to consider the indirect effects of livestock production on emissions of greenhouse gases. These are associated with the use of fossil energy for production and transport of feeds and chemical fertilizers. And with changes in the amount of C stored in the ecosystem (standing biomass and soil) as a consequence of changes in land use caused by the animal production system (*e.g.* deforestation for the production of feeds and forages). International organizations and governments are developing plans to reduce the emission of greenhouse gases.

3.6. ENVIRONMENTAL LEGISLATION

Like other European countries, The Netherlands developed legislation in the last decades of the 20th century to reduce nutrient emissions from agriculture [16, 61]. Initially, this legislation focused on different aspects of animal manure production and management. Most of the measures are based on the fact that livestock manures in The Netherlands generally are stored and applied as slurry, *i.e.* the mixture of faeces, urine and some cleaning water. Dutch legislation includes the following measures:

- Discharge of livestock effluents to surface waters has been prohibited since the 1960s. Related measures have been gradually tightened up to include run-off of dirty water from farm-yards, feedlots, hardstandings and stored manure.
- Since 1985, there have been several attempts to stop the growth of livestock numbers. Initially, farms with a manure production of more than 55 kg P ha^{-1} year^{-1} were not allowed to increase livestock density. Later on, the government established several schemes to reduce livestock numbers by buying pig and poultry production rights (rights expressed in production of manure P). This led to a reduction in manure production of 4.4 million kg P year^{-1} [62].
- Gradually increasing, P-based restrictions on the rate of manure application (Table 3.9). For grassland these rates included the P excreted by grazing animals. The very high values for grassland and maize in 1987 reflected the high livestock densities and common practices of manure disposal in some regions of the country. To comply with this measure, livestock farmers had to calculate the production of manure P (using standardized P excretion figures for different livestock categories) and to dispose of the manure surplus *via* the ‘manure bank’ or by means of a ‘manure transfer contract’ directly to arable farms or livestock farms with a low animal density. Since the introduction of the restrictions on the rate of manure application, manure disposal and acceptance are organized on a type of market, the ‘manure market’, where the livestock farmer has to pay for the disposal of the manure surplus, *viz.* transport costs and possibly a fee to the farmer who accepts the manure. Hence, these restrictions forced manure transport from farms with a high livestock density to farms with a low livestock density [17], as well as the adoption of animal feeding practices aiming at a reduction in P excretion (Chapter 4). They hardly affected livestock density.

- A ban on slurry application in the season without plant growth, hence from September 16th to January 31st, and later when the soil is frozen or covered with snow. This applies to all types of land use, except to arable land on clay and peat where application of animal manure in spring may cause serious damage to soil structure. As a consequence of this regulation, all livestock farms need to have slurry storage capacity for at least 5 months. Slurry silos must be covered to reduce ammonia volatilization.
- Slurry application techniques with low ammonia emission rates are compulsory on grassland and arable land on almost all soil types (Chapter 7). On grassland, deep injection, shallow injection, shallow injection with open slits, and application by trailing-feet machines are officially accepted as low-emission techniques. On arable land, direct incorporation of the slurry is required.
- Construction of so-called green-label livestock buildings is stimulated to reduce ammonia volatilization.
- Since 2002, livestock farms which produce more manure N than 170 kg ha^{-1} $year^{-1}$ for arable land and 250 kg ha^{-1} $year^{-1}$ for grassland, are obliged to enter into manure transfer contracts with other farmers, to reduce their livestock numbers, or to buy additional land. Manure transfer contracts can be made with arable farmers, other (less intensive) livestock farmers or manure processors [63].

TABLE 3.9. TIMETABLE FOR THE IMPLEMENTATION OF MAXIMUM ALLOWED RATES OF MANURE APPLICATION (IN KG P HA^{-1} $YEAR^{-1}$) FOR DIFFERENT CROPS IN THE NETHERLANDS.

Year	Grassland	Maize	Arable crops
1987	109	153	55
1991	87	109	55
1994	87	66	55
1996	59	48	48
1998[1]	52	44	44
2000[1]	37	37	37
2003[1]	35	35	35

[1] Only on small farms (< 3 ha or < 3 livestock units), which were exempted from the Mineral Accounting System (MINAS), introduced in 1998

The above-mentioned regulations, determining the rate, period and technique of slurry application, strongly improved the utilization efficiency of slurry nutrients [29, 64]. They make animal slurries a reliable source of plant nutrients and lead to lower nutrient losses if the application rates of artificial fertilizers are adjusted properly to take account of the increased availability of slurry nutrients. Since the introduction of the legislation described, large quantities of manure are transported from regions with a manure surplus to regions with capacity to apply more manure in the fertilization plan. This caused reductions in the use of chemical fertilizers (Table 3.1). However, these reductions were not sufficient to reduce nutrient losses to ecologically acceptable levels [*e.g.* 65]. Therefore, the Dutch Government introduced the Nutrient Accounting System (MINAS) [66]. Since 1998, N and P balances of individual farms serve as a basis to stimulate greater N and P use efficiency and to discourage excessive N and P use by financial penalties. Consequently, levy-free N and P surpluses have been set for grassland and arable land, which have been lowered gradually until 2004 (Table 3.10). The ultimate objective of MINAS is to guarantee a nitrate content in groundwater of less than 50 mg per litre (World Health Organisation's standard for drinking water, adopted in European legislation; [37, 38]), and a 50% reduction in N and P loads to surface waters.

TABLE 3.10. TIMETABLE FOR THE IMPLEMENTATION OF LEVY-FREE N AND P SURPLUSES (KG HA^{-1} $YEAR^{-1}$) FOR DIFFERENT TYPES OF LAND USE IN DUTCH AGRICULTURE. THE N AND P SURPLUSES ALLOWED TO A FARM ARE THE WEIGHTED SURPLUSES FOR THE GRASSLAND AND ARABLE CROPS ON THE FARM

Element and land use	1998	2000	2001	2002	2003	2004
N on grassland	300	275	250	220/190*	180/160*	180/140*
N on arable land, clay/peat	175	150	150	150	100	100
N on arable land, sand	175	150	125	110/100*	100/80*	100/60*
P on grassland+	17.5	15.3	15.3	10.9	8.7	8.7
P on arable land+	17.5	15.3	15.3	13.1	10.9	10.9

* Last figure refers to dry sandy soils; + fertiliser P is not yet included in the P balances

Table 3.10 shows the rapid decline in the levy-free surpluses. On dairy farms, for instance, the levy-free N surplus for grassland decreased from 300 kg ha^{-1} in 1998 to 180 kg ha^{-1} in 2004 and even to 140 kg ha^{-1} on dry sandy soils, where a relatively large part of the N surplus is lost by nitrate leaching. Meanwhile, the levy-free P surplus decreased from 17.5 to 8.7 kg P ha^{-1}. Levies gradually increased to 2.30 Euro per kg N and *ca*. 20 Euro per kg P in excess of the levy-free surpluses. Levies apply to the N and P excesses of the whole farm. Hence, a 40-ha farm with an N surplus of 25 kg ha^{-1} $year^{-1}$ in excess of the levy-free surplus has to pay 2300 Euro (40*25*2.30).

Between 2001 and 2005, almost all Dutch farmers had to submit an annual MINAS declaration. Only very small farms were exempted from this; they had to observe the maximum rates of manure application, presented in Table 3.9. On the MINAS declaration form, farmers had to quantify N and P inputs (from outside the farm) *via* animal manure, sewage sludge, compost, soil, inorganic fertilizers, concentrates, roughage, animals and biological N fixation by legumes (except N fixation by clovers in grassland) and N and P outputs (from the farm) *via* crops, animals, animal products, animal manure and roughage. According to the MINAS guidelines, the MINAS-N balances did not include N inputs *via* atmospheric deposition and the MINAS-P balances did not include P inputs *via* chemical fertilizers. In addition to the N outputs mentioned, the guidelines allowed to include some extra N output on farms with a high livestock density, to account for gaseous N losses from manure that supposedly had to be exported from the farm. Manure and feeds imported to and exported from the farm had to be weighed and analysed and other inputs and outputs should be traceable in the financial administration of the farm. In some cases, fixed values for inputs and outputs could be used on the declaration, but this was discouraged. MINAS declaration forms had to be submitted to a specialized office of the Ministry of Agriculture, Nature and Food Quality, where they were verified and possible levies were calculated and charged. In the first years after the introduction of MINAS, a considerable number of farmers had to pay levies [40]. In some livestock sectors, about 40% of the farmers exceeded the levy-free surpluses. This shows that farmers had problems to adapt farm management to the annually changing levy-free surpluses (Table 3.10). In fact, they had to consider two elements, N and P, and for each the difference between several inputs and outputs.

Despite the problems mentioned, the high levies forced farmers to reduce the N and P surpluses to the required level and to adapt N and P management accordingly within a few years. For instance, the average N surplus of specialized dairy farms, calculated according to the MINAS procedures, was *ca*. 300 kg ha^{-1} $year^{-1}$ in the period 1996–1998 [24, 67] and *ca*. 165 kg ha^{-1} $year^{-1}$ in 2004. The most suitable measure to reduce the N surplus of those farms to the permitted level was a reduction in N inputs *via* pig manure and chemical fertilizer. This possibly caused a small reduction in herbage dry matter and protein yield. On some farms it was possible to compensate this with higher feed purchases, but on other farms the limit on the P surplus prevented this. The average P surplus of specialized dairy farms in 1996–1998 was *ca.* 4 kg ha^{-1} year^{-1} higher than the levy-free surplus in 2004 [67]. Other possibilities to cope with the required reductions in N and P surpluses on the dairy farms were: improvement of N and P management in different parts of the production system, manure export to another farm, and buying additional land (extensification of the farm). The measures actually chosen depended on the economic situation and skills of the farmer.

MINAS caused a significant improvement of N and P management in Dutch agriculture. This is illustrated for a specialized dairy farm in Figure 3.1 and a mixed dairy + pig farm in Figure 3.2 [68]. Both farms are situated on sand in the Province of Gelderland and participated in a regional study on ammonia volatilization [69]. The Figures 3.1 and 3.2 are outputs of calculations with the model FARMMIN of the annual N flows and losses on theses farms [70]. The calculations apply to 2002.

The specialized dairy farm (Figure 3.1) had 68 dairy cows + young stock on *ca*. 41 ha of grassland and 6 ha of silage maize. Annual milk production was 12,870 kg ha^{-1} and 8,830 kg cow^{-1}. The N surplus in 2002, calculated according to MINAS, was 163 kg ha^{-1}, whereas the P balance of this farm showed a deficiency of 3 kg P ha^{-1}. These figures differ strongly from those presented in Table 3.7 and show the effects of legislation on N and P management on this type of farms. Calculated annual N losses from this farm were: 58 kg N ha^{-1} by ammonia volatilization and 5.0 kg N ha^{-1} by emission of nitrous oxide. The calculated amount of residual inorganic N in the soil profile in autumn was 50 kg ha^{-1} year^{-1}. This was lost in the following winter by denitrification and nitrate leaching, causing an average nitrate concentration of 51 mg per litre in the groundwater on this farm. This is slightly higher than the standard for drinking water (Table 3.8). A small reduction in the rate of fertilizer N would be sufficient to attain the standard for drinking water.

The mixed dairy + pig farm (Figure 3.2) had 24 dairy cows + young stock on 15 ha of grassland and 2 ha of silage maize. In addition, the farm had a unit of 43 sows and was selling the piglets at a weight of *ca*. 25 kg to a growing-finishing farm. Annual milk production amounted to 11,420 kg ha^{-1} and 8,040 kg cow^{-1}. The MINAS-N surplus was 209 kg ha^{-1} in 2002, whereas the MINAS-P surplus was 10 kg ha^{-1}. The levy-free surpluses for this farm were 231 kg N ha^{-1} (including the allowed correction for gaseous N losses) and 11 kg P ha^{-1}. Calculated annual N losses from this farm were: 66 kg N ha^{-1} by ammonia volatilization and 6.5 kg N ha^{-1} by emission of nitrous oxide. The calculated amount of residual inorganic N in the soil profile was 60 kg ha^{-1}, causing an average nitrate concentration of 62 mg per litre in the upper groundwater. This farm will have more problems than the specialized dairy farm, described in Figure 3.1, to meet the requirements of future environmental legislation. The plan of the Dutch government to reach P equilibrium on agricultural soils in 2015 will force this farm to enter into a manure transfer contract in the near future. In addition, the rate of N application has to be reduced to reduce nitrate leaching.

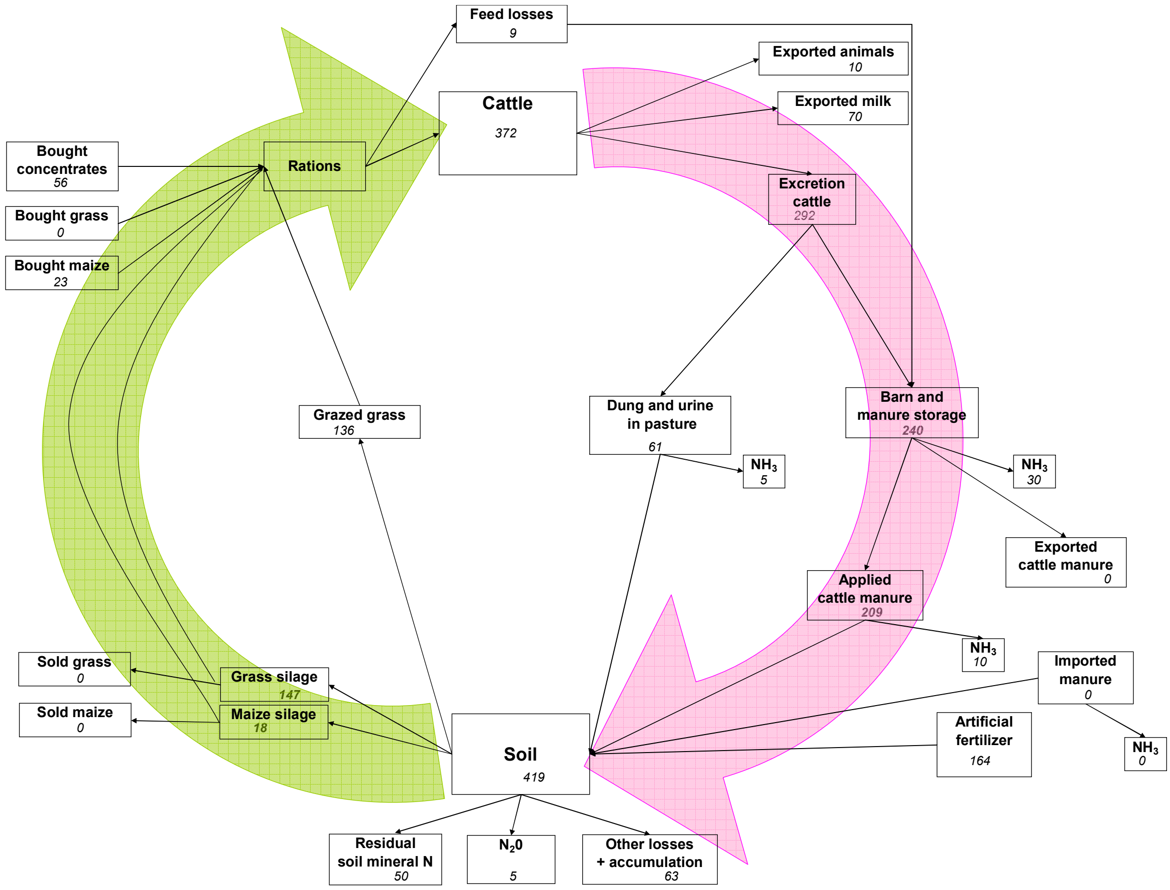

FIG. 3.1. Calculated N flows and losses in 2002 on a specialized dairy farm on a sandy soil in the Province of Gelderland, The Netherlands. Calculations have been made with the simulation model FARMMIN [70]. Figures are kg N ha^{-1} year^{-1}.

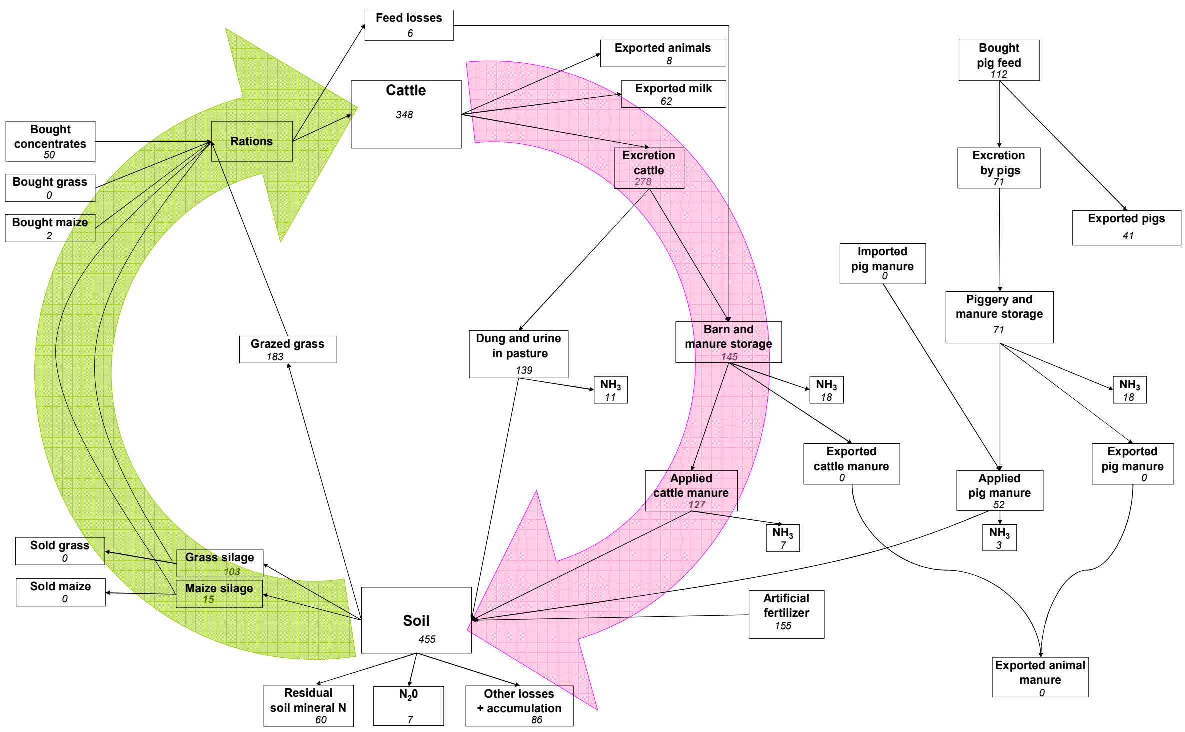

FIG. 3.2. Calculated N flows and losses in 2002 on a mixed dairy + pig farm on a sandy soil in the Province of Gelderland, The Netherlands. Calculations have been made with the simulation model FARMMIN [70]. Figures are kg N ha^{-1} year^{-1}

The Nitrate Directive (91/676/CEC), issued in 1991 by the Commission of the European Communities, was an important step towards a common European approach concerning the protection of surface and groundwaters against pollution by nitrates from agriculture [38]. The objective of the Nitrate Directive is to reduce water pollution caused, or induced, by nitrates from agricultural sources, as well as to prevent future pollution. It applies to (1) surface freshwater and groundwater used, or intended for the abstraction of drinking water, and (2) natural freshwater bodies and coastal and marine waters which are, or may become, eutrophic. The Nitrate Directive invites the governments of the Member States of the European Union to identify zones, which drain into waters (potentially) affected by pollution with nitrates, and to establish an action programme for these zones. This action programmemust include measures related to the period and rate of application of animal manures and chemical fertilizers, and to the storage capacity for animal manures. Member States should also establish codes of good agricultural practice, to be implemented by farmers on a voluntary base, and to contain provisions for environmentally friendly storage and application of animal manures and chemical fertilizers. An important feature of the Nitrate Directive is that it specifies the maximum amount of animal manure that can be applied to farmland each year. This should not exceed 170 kg total N ha^{-1} $year^{-1}$, including excreted N by grazing livestock. A derogation may be approved for crops with a long growing season and a large capacity for N uptake and for conditions with a large denitrification. In many parts of Western Europe, grass has a long growing season as well as a large capacity for N uptake. Therefore, the European Union approved a derogation for grassland in The Netherlands, allowing a maximum application of manure N of 250 kg ha^{-1} $year^{-1}$.

Despite the good effects of MINAS on environmental quality in The Netherlands, the European authorities did not accept MINAS as the Dutch method to comply with the Nitrate Directive. Therefore, the Dutch government enacted new legislation, which is in force since 2006, determining maximum rates of application of:

- manure total N. This is 170 kg ha^{-1} $year^{-1}$ for arable crops and 250 kg ha^{-1} $year^{-1}$ for grassland-based livestock farms (farms with > 70% grassland).
- effective N (fertilizer equivalents = fertilizer N + manure N * efficiency index) (Chapter 7). High estimates of the efficiency indices have been included in legislation to stimulate effective use of manure N. In principle, the maximum rates of effective N are based on the fertilizer recommendations for the different crops, but lower values have been set for crops where these recommendations are considered too high to reach the required water quality.
- total P (manure P + fertilizer P). According to the Fourth National Environmental Policy Plan, the allowed maximum rates of P application will be reduced gradually to reach P equilibrium in 2015 [36].

3.7. A SYSTEMS APPROACH TO SUSTAINABLE MANURE MANAGEMENT

Livestock production has expanded rapidly in many Asian countries and this expansion is expected to continue. Farm animals consume forages and concentrates (energy, proteins, minerals, and vitamins) to produce meat, milk and eggs. Manure is an inevitable by-product of this process. Manure contains the undigested fraction of the organic matter in the diet, and generally > 70% of dietary N, and > 65% of dietary P (Chapter 4). Hence, manure is an important source of organic matter for agricultural soils and may provide a major contribution to biological, physical and chemical soil quality. In addition, it is an important source of plant nutrients.

Farm animals concentrate organic matter, plant nutrients, and harmful constituents in the places where they are kept. Generally, this causes problems of nutrient depletion and soil degradation in the regions where the feed is produced (unless manures are returned), and nutrient accumulation and environmental pollution in the regions where livestock production is concentrated (unless manures are removed). The higher the livestock density, the bigger these problems tend to be. Land application to fertilize crops is considered the most suitable method to dispose of or to utilize animal manures. This requires much attention in Asian countries where the questions need to be answered on the soil types and crops that respond well to applications of animal manure. The best responding soils generally are those with a low natural fertility. Such soils may be improved significantly by regular additions of animal manure. An example of this can be seen in The Netherlands, where originally very poor sandy soils actually support very prosperous livestock and crop production systems.

Present-day production of the most important arable crops in Asian countries (rice, wheat, maize) is largely based on chemical fertilizers. This can be concluded from reports on long-term experiments with rice in monoculture and rice-wheat rotations [71, 72]. Many of these experiments show a decline in yield and soil fertility over time [71]. The following possible causes have been mentioned [73]:

- A decrease in soil organic matter content and related decline in physical soil quality and soil N supply (particularly in rain-fed conditions).
- A decline in soil N supply due to changes in the composition of soil organic matter associated with prolonged periods with anaerobic soil conditions.
- Negative balances of P, K and other secondary and micronutrients. This is often a problem in crop production systems based on chemical fertilizers. In these systems, N receives most attention and supply of other nutrients may be neglected.

Livestock manures may play an important part in maintaining high levels of crop production. Crop nutrition research should consider effective utilization of this local resource. This requires a systems (holistic) approach to agricultural development in a region. Although European agriculture has useful expertise and experiences, research will be necessary to develop sustainable local systems in Asian countries. The productive tropical and subtropical environments in Southeastern Asia appear very promising for the development of integrated agricultural production systems with crops, livestock, aquatic products, mushrooms, *etc*., and for effective use of land, water, solar and fossil energy, and plant nutrients.

Promising systems of manure utilization were observed on the small dairy farms we visited in the neighbourhood of Ho Chi Minh City, Viet Nam. Although these farms had limited land available, the farmers were growing productive tropical grasses (*Pennisetum purpureum, Panicum maximum, Brachiaria ruziziensis*), and fertilizing them with liquid manure. Such grasses require large amounts of plant nutrients and may be able to utilize large rates of animal manure. Besides, we were told that manure solids were in high demand for coffee, pepper, and other high-value crops. Nutrient use efficiency of these farms may be assessed by means of farm-gate nutrient balances. It will be very useful if local researchers draw up nutrient balances of these farms. This will provide relevant local information on nutrient use efficiency and contribute to the development of sustainable livestock and crop production systems.

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4. PRODUCTION AND COMPOSITION OF MANURE

The main factor that influences the total amount of N and P in animal excreta is diet. In fact, 55–90% of the N and P content of animal feed is excreted in faeces and urine [1, 2]; (Table 4.1). Within species, the exact proportion excreted varies according to a range of factors including diet composition, animal performance (*e.g.* amount of milk produced, liveweight gain), size, age, sex and husbandry [3]. It is clear that diet composition, feed conversion and animal size and performance are the factors with the most important impact on manure production and composition.

TABLE 4.1. PERCENTAGE OF DIETARY N AND P EXCRETED BY LIVESTOCK [1, 2]

Animal category	N excretion (% of intake)		P excretion (% of intake)
	Ryser *et al.* [1]	Tamminga *et al.* [2]	Ryser *et al.* [1]
Dairy cow	65–80	79	65–80
Growing cattle (beef)	75–80	77–89[1]	70–85
Sow with piglets	75–80	76	75–85
Growing-finishing pig	70–80	66	75–85
Laying hen	65–80	65	85–90
Broiler	55–65	56	50–65

[1] Lower values for steers (3–16 months), higher values for suckler cows

The data from [2] in Table 4.1 are estimates of the average situation on commercial farms in The Netherlands in 1998, based on statistical information, on-farm research and expert knowledge. These results have been included in environmental legislation.

4.1. ESTIMATION OF N AND P EXCRETION BY FARM LIVESTOCK

A number of methods exist for quantifying nutrient excretion by farm livestock. These include:

- direct measurements with livestock;
- direct measurements with manure;
- input-output measurements.

Direct measurements with livestock may provide the most accurate measure of nutrient excreted, but require either total collection of faeces and urine or reliable markers for spot sampling. This is an expensive and time-consuming method, and the values obtained can only be applied to similar types of livestock (*e.g.* breed, age, sex, growth rate) and diets.

Estimates of nutrients excreted in manure by direct measurements and analysis of the manure may be achieved at less cost than direct measurements with livestock (in terms of the number of samples and analyses required). However, the amounts of manure produced are difficult to quantify, and obtaining representative samples for analysis can be particularly difficult. This approach also suffers from the fact that the results obtained are only applicable to the particular factors and conditions prevailing during the period of observations and sampling. In the case of N, N losses *via* gaseous NH_3 emission occur very rapidly following excretion and need to be considered in relation to the point of assessment and to what extent NH_3 may impact on the measurements or estimates of manure N output.

A more common approach to estimating nutrient excretion by farm livestock is to assume that the amount of nutrients excreted in faeces and urine is the total amount consumed minus the nutrient content of products (*e.g.* milk, eggs, liveweight gain) [4]. For the purpose of the example, P is used, and this approach may be represented as:

$$P_{\text{excreted}} = P_{\text{intake}} - P_{\text{animal products}} \quad \textbf{(1)}$$

This approach may be applied at the level of an individual animal, farm or region. It requires information on:

(a) P intake, and
(b) the P content of animal products.

4.1.1. Estimating P intake

Estimating P intake requires information on the feed consumed by different classes of livestock and the P content of the feed. For some species, *e.g.* growing-finishing pigs, laying hens and broilers, feed intake is generally well known. For others, non-confined animals and particularly ruminants, both the amount and P content of some or all of the feed consumed may be unknown. One approach is to assume that animals consume sufficient feed to meet their energy requirements[1]. If this is the case, then the intake of feed may be described as follows:

Energy intake (MJ) = Energy (MJ) required for maintenance + production **(2)**

In many countries data exist to calculate the energy required for maintenance and production and, therefore, provide an estimate of energy intake. With this information, and information on the energy content of feeds available, it is possible to calculate likely intakes of different feeds:

Energy intake (MJ) = energy (MJ) in forage crops (fresh or conserved) + energy (MJ) in purchased concentrate feeds **(3)**

While the supply of purchased concentrate feeds is usually controlled and understood, intake of forage crops is less well known. Therefore, equation **(3)** can be re-written as:

Energy intake (MJ) – energy in purchased concentrate feeds (MJ) = energy in forage crops consumed (MJ) **(4)**

Dividing the energy supplied from forage crops (MJ) by the energy concentration of the forage(s) consumed (MJ/kg DM), it is possible to arrive at an estimate of the DM intake of livestock. The P intake can then be estimated from this.

[1] On diets that are not deficient in any of the major nutrients, the need to meet energy requirements usually determines intake.

This approach has a number of limitations:

- It assumes that the energy requirements are well known (also energy content and utilization). There is some consensus between experts for many estimates of maintenance and production, but not for all.
- It can only be applied when there is not more than one feed for which the intake is unknown.
- It assumes that animals are eating to meet their energy requirements for maintenance and production. While this is generally the case, in situations where the diet is deficient in one or more essential nutrients, *e.g.* protein or P, then energy intake may be inadequate and both DM intake and production will decline.
- It assumes that the energy content of the forage and purchased concentrate feeds are known. The energy value of feeds (and particularly forages) can vary significantly, depending on feed type, weather conditions during growth, maturity at harvesting, and processing or conservation method. Rapid and reliable methods of predicting the energy content of many of the feeds consumed by ruminant livestock are not widely available, and the use of standard values may lead to erroneous conclusions in some estimates.
- It assumes that the P contents of the feeds are known. As discussed above, P contents can vary even within feed classes.

4.1.2. The P content in animal products

While there is some variation in the P content of milk, meat and eggs, differences are generally small (particularly in relation to differences in P content of feed). For example, the P content of whole cow milk produced during the winter and summer months are given as 96 and 93 mg/100 ml [5]. However, such differences are negligible in the calculation of P excretion according to equation **(1)**. Therefore, it is possible to apply standard values for N and P contents in live animals and animal products. The following data, taken from the Mineral Accounting System (MINAS) in The Netherlands, can also be used for calculations of N and P excretion in Asian production systems (Table 4.2).

TABLE 4.2. NITROGEN AND P CONTENTS OF LIVESTOCK (G OF N AND P PER KG LIVE WEIGHT) AND LIVESTOCK PRODUCTS (G OF N AND P PER KG PRODUCT). SOURCE: [6]

Animal, animal product	g N kg^{-1}	g P kg^{-1}
Cow milk	5.4	0.92
Calf, lean beef animals	29.4	7.6
Dairy cow	25.6	7.4
Young stock, dairy breeds	25.6	7.4
Sheep	25.0	6.0
Goat	24.0	6.0
Piglet, at weaning	24.0	5.2
Slaughter pig	24.8	5.0
Sow	25.5	5.0
Eggs	19.2	2.1
Broiler chicken	28.0	4.7
Laying hen	28.0	3.1
Duck	25.9	5.7
Turkey	33.0	7.2

In summary, the method as described above is a well-recognized procedure for estimating the excretion of nutrients by farm livestock. It has been adopted for developing input/output relationships for nutrients, particularly N and P, and has the advantage that it can be applied at a farm or regional level based on local or generalized information. It uses data that are, in most cases, readily available and is recommended as a methodology for estimating N and P excretion by livestock. Calculations of N and P excretion by farm livestock in this way have been applied in several European countries [2, 3, 7] and provide a standard approach for estimating equivalent data for N and P in Asian countries, where few relevant estimates of any sort exist, to date. A format to calculate N and P excretion in livestock is shown in Annex Tables A1–A5, showing example calculations for dairy cows, sows + piglets, growing-finishing pigs and laying hens. These calculations are facilitated using a simple EXCEL spreadsheet following the format shown [3].

A tentative approach has been adopted in some recent regional studies with nutrient balances, estimated on a national basis, which has provided estimates of livestock manure nutrient outputs in Asian countries [8]. These have been based on likely inputs and animal performance for different production intensity classes of 12 livestock categories [9]. In the absence of relevant measurements, it is suggested that these data, as summarized in Table 4.3, might be used as a first estimate of likely outputs. For comparison, average N and P excretion data for the most important livestock categories in The Netherlands in 1998 have been included ([2]; Bannink and Valk, unpublished). These data are based on statistical information on animal numbers, animal production, consumption and composition of concentrates and forages, as well as on-farm research and expert knowledge. Characteristic performance parameters of Dutch production systems are added for reference (derived from [2]).

For the highest production intensity of each livestock category in Asia, the annual excretion of N, P and K per animal place[2] was estimated on the basis of experience from Thailand, China, Denmark and Switzerland [8]. For the lower intensity classes, it was estimated how much lower excretions would be as compared to the highest class, based on live weight, production and feed quality.

It is remarkable that P excretions for intensity class 1 in Asia are similar to average P excretions in The Netherlands, whereas the corresponding N excretions in Asia are much lower, except for laying hens and broilers (Table 4.3). The causes of these differences are not clear. Dutch livestock farmers were already stimulated by environmental legislation to reduce N and P excretion of farm animals in 1998, so it is unlikely that utilization of dietary N was much more efficient on intensive livestock farms in Asia than in The Netherlands. This suggests that N excretion values for livestock in intensive production systems in Asia are under-estimated. The large differences in N and P excretion between intensity classes should also be considered with caution (Table 4.3). The data presented indicate an almost similar excretion of N and P per kg of animal product for the different intensity classes. That is questionable and points to the need of local on-farm studies using input data for the nutrient balance calculations, as outlined above. As indicated above, feeding practice has greatest impact on manure N and P outputs. Therefore, it is important that production intensity is considered for each country and even for distinct regions in large countries; a standard value for Asia being clearly inappropriate.

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[2] In Europe, N and P excretions are generally expressed per animal place per year. This is particularly important for animal categories with several production rounds per year. For instance, an animal place for growing-finishing pigs in The Netherlands has 3.12 production cycles per year (Table 4.3). Hence, the excretion data relate to the production of 3.12 pigs for slaughter (starting and slaughtering weights 25 and 114 kg, respectively).

TABLE 4.3. ESTIMATES OF N AND P EXCRETION OF DIFFERENT ANIMAL CATEGORIES AND PRODUCTION INTENSITIES IN ASIAN COUNTRIES IN 2000 [8]. FOR COMPARISON, AVERAGE N AND P EXCRETION PER ANIMAL CATEGORY IN THE NETHERLANDS IN 1998 ([2]; BANNINK AND VALK, UNPUBLISHED). DATA: KG (ANIMAL PLACE)$^{-1}$ YEAR^{-1}

Animal category	Intensity class[1]	Live weight (kg)[2]	Excretion / animal place / year N (kg)	P (kg)
Asia				
Dairy cows[3]	1	600	100	17.5
	2	500	65	11.3
	3	400	26	4.5
	4	300	6.5	1.13
Young dairy stock	1	300	40.0	7.0
	2	250	32.5	5.7
	3	200	14.3	2.5
	4	150	3.9	0.7
Pigs	1	31–73[4]	8.4	2.3
	2	39–61[4]	7.3	2.1
	3	26–65[4]	4.9	1.2
Laying hens (10 birds)	1	21	6.5	1.9
	2	21	5.6	1.5
	3	21	4.6	1.1
Broilers (10 birds)	1	10	5.3	1.1
	2	10	4.6	0.9
Turkeys		5.0	1.0	0.22
Ducks		2.0	0.6	0.13
Netherlands				
Dairy cows[5]	Average (intensive)	550–600	141	17.9
Young dairy stock	ditto	40–550	69	7.4
Sow + piglets (25 kg)[6]	ditto	132–205	29.5	6.4
Growing-finishing pigs[7]	ditto	25–114	13.4	2.1
Laying hens (10 birds)[8]	ditto	13.2–19.0	6.84	1.8
Broilers (10 birds)[9]	ditto	0.42–19.6	5.84	0.96
Turkeys[10]	ditto	0.057–14.0	1.92	0.34
Ducks[11]	ditto	0.053–3.0	1.01	0.20

[1] Production intensity classes are defined on the basis of indicators such as average carcass weight, the number of slaughtered animals per number of total stock, and the milk yield per cow; intensity classes vary across animal categories within countries [8]; [2] Live weights in Asia apparently are averages during the production cycle; those in The Netherlands are ranges from the start to the end of a production cycle [2]; [3] Average milk yield for intensity classes 1, 2, 3 and 4: 7000, 3500, 1700 and 500 kg cow^{-1} year^{-1}; [4] Average live weights determined individually for each country; [5] Average milk production 6816 kg cow^{-1} year^{-1} (= 7225 kg fat-corrected milk). Diet: fresh grass, grass silage, maize silage, concentrates; [6] One sow gives birth to 25.5 life piglets per year; these piglets go to a growing/finishing operation after 75 days at a weight of *ca*. 25 kg (excretions are for sow + piglets); [7] Fattening pigs grow in 117 days from 25 kg to a slaughter weight of 114 kg; feed conversion = 2.73 kg kg^{-1}; 3.12 rounds per year (no empty period); [8] Hens in battery cages produce 20.9 kg eggs in a production period of 418 days; feed conversion = 2.29 kg kg^{-1}. Performance in other housing or in free-range systems is lower and excretions higher; [9] Broilers grow in 42 days from 42 g to a slaughter weight of 1960 g; feed conversion = 1.82 kg kg^{-1}; [10] Turkeys grow in 132 days from 57 g to a slaughter weight of 14,000 g (average of males and females); feed conversion = 2.65 kg kg^{-1}; [11] Ducks grow in 45 days from 53 g to a slaughter weight of 3000 g; feed conversion = 2.45 kg kg^{-1}.

In summary, most of the N and P excretion data for Asian livestock production systems, as presented in Table 4.3, appear low compared to European experience.

4.2. NUTRIENT CONTENT OF FARM LIVESTOCK MANURES

It is generally accepted that animal manures are valuable sources of nutrients and organic matter for use in the maintenance of soil fertility and crop production. However, for reliable fertilizer planning and to allow confidence amongst farmers in the use of manures as nutrient sources, it is necessary to know the nutrient content of the manures. Data on typical nutrient contents for manure of the major livestock categories are available from many different countries and have been summarized in Table 4.4, including some data on piggery waste water and solids from studies in Thailand and Singapore [10, 11]. At the present time, other data of this type is lacking for Asian countries. These figures can serve as an initial guide, at least for those interested in nutrient recycling within crop production systems. Of course, Asian information is much too limited and European information is likely to differ too much from that for Asian countries, for these data to be used without local manure sampling and analysis. There is an urgent need to initiate a programme for analysing livestock manures in Asian countries and this needs to follow the development of robust sampling and analysis methods appropriate for Asian livestock production systems. Whilst it can be seen that, where data for different manure types are available from different sources, there is often close agreement in the values (Table 4.4), the ranges are also considerable, reflecting large differences between farms and regions. Comparable systems (building, animal nutrition and management, manure management) will have much smaller ranges in values and it is useful to try to establish average compositions for different conditions. More detailed comments on the importance of manure nutrient content and the need for representative sampling and analysis follow in Chapter 7.

Animal slurry is the mixture of faeces, urine, water (spilt drinking water, cleaning water, rain), and sometimes some feed residues and bedding material. The large differences in DM and nutrients content, as shown in Table 4.4, are mainly caused by different quantities of water being added. The lowest dry matter (DM) contents in slurries in Europe are comparable to the DM content of waste waters in Asia. Many European farmers limit additions of water to slurry to reduce costs of slurry storage and handling. Often, it is useful to calculate and compare nutrient contents in manure on the basis of manure DM. In this way a check on slurry DM made using a hydrometer, or other simple device to measure specific gravity, can provide a useful means of adjusting the nutrient content of a slurry on the basis of a laboratory report on a sample taken previously from the same farm (see Chapter 7).

Solid livestock manure in Europe generally is the mixture of faeces, (part of the) urine, feed residues and bedding material (straw). Solid manure of laying hens mainly consists of naturally or artificially dried droppings. Solid manures are often composted before land application. This process causes large gaseous losses of N by NH_3 volatilization and nitrification/denitrification. This decreases the fraction of readily available N (NH_4^+-N) and is the main cause of the lower N/P and N/K ratios generally observed in solid manure compared to slurry (Table 4.4).

TABLE 4.4. AVERAGE AND RANGE OF COMPOSITION VALUES (KG TON^{-1}) FOR DIFFERENT TYPES OF MANURE AND EFFLUENT REPORTED FROM DIFFERENT SOURCES

Type of manure		Dry matter (kg ton^{-1})	N	NH_4^+-N	P	K	Mg	Source
Slurry								
Pigs	average	51	4.8	3.5	0.9	2.7	0.6	1
	range	15–92	1.2–8.2	1.9–6.1	0.13–2.2	0.5–6.6	0.1–1.8	1
Sows + piglets	average	50	4.2	2.5	1.3	3.6	0.66	2
Fattening pigs	average	90	7.2	4.2	1.8	6.0	1.08	2
Poultry	average	170	11.1	5.2	3.9	4.4	1.7	1
	range	10–300	2–18	1.9–7.8	0.39–6.5	2.1–7.5	0.2–3.6	1
Laying hens	average	145	10.2	5.8	3.4	5.3	1.3	2
Cattle	average	60	3.0	1.5	0.5	2.9	0.4	3
Cattle	average	86	4.4	2.2	0.7	5.1	0.78	2
Solid manure								
Pigs	average	243	6.9	2.2	2.4	5.4	1.6	1
	range	150–330	3.5–11	0.5–6.0	0.74–6.5	2.3–13.3	0.9–2.5	1
Pigs (+straw)	average	230	7.5	1.5	3.9	2.9	1.5	2
Laying hens	average	406	23.6	10.9	7.2	8.9	3.1	4
	range	220–550	5.1–25	37–60	3.5–11.8	5.0–12.5	1.2–6.0	4
Laying hens	average	515	24.1	2.4	8.2	18.8	2.9	2
Broilers	average	603	24.5	8.0	8.1	14.2	4.2	4
	range	450–850	21.8–40	2.0–15	3.0–10.9	5.6–19.1	2.5–6.5	4
Broiler litter	average	605	30.5	5.5	7.4	18.7	3.9	2
Cattle FYM	average	250	6.0	0.6–1.5	1.5	6.6	0.4	3
Cattle FYM	average	248	6.4	1.2	1.8	7.3	1.3	2
Waste water and solid manure (Asia)								
Pig waste water	average	12.5	1.55	0.34	0.40	0.67	0.04	5
Pig manure solids	average	-	2.70	1.44	3.8	0.70	1.6	5
Pig waste water (Singapore)	average	18.3	1.10	-	0.19	0.18	-	6

Sources:
1. Data from different countries within the MATRESA project [12].
2. Typical nutrient contents from a large database of analyses in The Netherlands [13].
3. Typical nutrient contents from a large database of analyses in England and Wales [14].
4. Data from several countries contributing to the RAMIRAN database on solid manures RAMIRAM: Research Network on Recycling of Agricultural, Municipal and Industrial Residues in Agriculture, sponsored by the FAO (http://www.ramiran.net/) [15].
5. Average from four farms in study on effluent handling and treatment in Thailand [10].
6. Estimated composition, assuming a production of 4.5 kg manure and a water use for cleaning the building and cooling the animals of 20 litres per standing pig population (SPP ≈ 54 kg) per day [11].

4.3. IMPACT OF DIET ON EXCRETION AND NUTRIENT CONTENT OF LIVESTOCK MANURES

In areas with a high livestock density (and high costs of sustainable manure management), it is important to limit the production of manure N and P as much as possible. Utilization of N and P by livestock and poultry is influenced by many factors associated with the feeds themselves, the method of feeding and the performance of the animals to which they are fed. Undigested N and P are excreted, by definition, in faeces; digested N and P, which is not utilized by the animal (*i.e.* retained in animal products), is excreted in urine. In view of their different digestive and metabolic processes, the potential for reducing N and P excretion by ruminants, non-ruminants and poultry are considered separately.

4.3.1. Non-ruminants - Nitrogen

The protein requirements of pigs and poultry for maintenance, growth, reproduction and (in the case of poultry) egg production have often been established by national bodies such as AFRC (UK), NRC (USA) and INRA (France) and these are used widely for formulating diets. In addition to total protein, requirements are given for essential amino acids. There is considerable genetic variation in growth characteristics of animals, particularly in the retention of protein in relation to energy deposition. As a result, the amino acid requirements of modern strains of pig and poultry may differ significantly from those used to develop 'national' standards. Each strain may have its own protein deposition level and, as a consequence, its own optimum amino acid level in the diet. It is not uncommon therefore for poultry or pig breeding companies to develop their own standards, which may be confidential and only available to those producers purchasing their livestock.

In order to achieve target levels of production, nutritionists also need to take account of the variability in the composition of feeds. The range of concentrations of lysine and methionine for a number of commonly used feeds is illustrated in Table 4.5.

TABLE 4.5. CONCENTRATIONS (G KG^{-1}) OF LYSINE AND METHIONINE IN FEED MATERIALS [16]

Feed material	Lysine (total)				Methionine			
	Mean	Min.	Max.	n =	Mean	Min.	Max.	n =
Barley grain	4.9	4.0	7.0	20	2.8	1.0	5.0	20
Wheat	3.8	3.0	4.7	19	2.9	1.0	5.0	19
Soyabean meal	34.0	27.0	36.5	6	6.9	6.5	7.3	6
Rapeseed meal	21.9	20.5	23.3	8	7.2	6.3	8.6	8
Fish meal	55.6	46.4	60.8	5	18.3	17.0	19.6	5

In addition to the total amount of amino acids present, the digestibility of amino acids within feed types can vary significantly, being influenced by growing, harvesting and processing systems; processing temperature in particular can affect amino acid digestibility significantly.

Although considerable variability exists between batches of the same feed, the logistics of feed storage, handling and manufacture mean that there is usually insufficient time to have feeds analysed for their composition. Feed formulators therefore tend to adopt a precautionary approach, with the result that the actual concentration of an essential amino acid may be both higher than anticipated and necessary for the target level of production.

Strategies that have been developed to reduce the surplus N related to production have recently been reviewed [17], most of which relate to the amount of N consumed relative to the output of the animal. These include the following:

- *Reducing surplus N intake by selecting appropriate feeds and reducing the safety margin.* An important problem in Europe is that the cheapest feeds (concentrate ingredients) are selected, and this often causes considerable surpluses of N in livestock diets. Limiting N (and P) intake as much as possible has not been an objective in the preparation of livestock diets. Recently, this is changing as a result of environmental legislation.
- *Selection of feeds with high N availability.*
- *Optimizing dietary amino acid balance.*
- *Formulation of diets to an ideal protein using synthetic amino acids* [18].
- *Inclusion of fermentable carbohydrates.* These affect the partitioning of N excretion between faeces and urine and reduce excretion of urinary urea, slurry pH and NH_3 emission [19].

In addition, a number of production-related strategies have been identified.

- *Phase feeding.* More precise targeting of diet to requirements has been shown to reduce N inputs and excretion. Pigs reared on a three-feed system excreted 8% less N over their lifetime than those on a two-phase system (Henry and Dourmad, 1993, cited in [17]).
- *Output optimization.* The law of diminishing returns applies to pig and poultry production as much as to other livestock systems, and the most economical weight gain may not occur when N utilization is maximized. Moreover, a reduction in dietary protein content may not result in lower N excretion per unit production if animals have to be retained longer as a result of lower growth rates.

An illustrative study on the effects of dietary protein content on pig performance, production and composition of faeces and urine, and NH_3 losses has been reported in [18].

4.3.2. Non-ruminants - Phosphorus

Plants store up to 80% of their seed phosphate as phytic acid. While plants have a natural mechanism to release the P when required for growth, pigs and poultry do not and, as a result, P in this form is not well utilized by non-ruminants. Traditionally, the P requirements of pigs and poultry have been met by supplementing with inorganic P, with a large safety margin. The non-digested phytic-P and any excess supplementary P are excreted in the faeces. Because some feedstuffs are high in phytate, and because there is some endogenous phytase in certain small grains (wheat, rye, triticale, barley), there is a wide variation in the bio-availability of P in feed ingredients. For example, the P in maize grain is only 12% available while the P in wheat is 50% available. The P in dehulled soybean meal is more available than the P in cottonseed meal (23 *vs*. 1%), but neither source of P is as highly available as the P in fishmeal (93%) or dicalcium phosphate (90–95%). Similarly, P availability in different forms of dicalcium phosphate can vary from 89 to 100% compared to monocalcium phosphate [20]. The choice of supplementary P can therefore have a significant impact on the amount of P excreted by pigs and poultry.

Environmental concerns have led to reductions in dietary P content of pig and poultry diets in developed countries. These have been achieved partly through:

- a better understanding of requirements and P availability in feeds (discussed above);
- phase feeding, to match supply and requirements more closely;

- the use of exogenous phytase enzymes, generally of fungal origin;
- limiting the safety margins;
- good animal management.

Supplementing the diet with the enzyme phytase is an effective means of increasing the breakdown of phytate P in the digestive tract and allows the use of inorganic P supplements and P excretion in faeces to be reduced. For poultry also, it has been suggested that P excretion can be reduced by up to 70%, and manure volume by up to 14%, as a result of using dietary phytase [21]. The extent to which these enzymes are likely to be used by the pig and poultry industry will be influenced largely by the cost of the enzyme relative to the cost of supplementing diets with inorganic P. Public pressure on producers (legislation) to reduce P losses to water resources is also important.

4.3.3. Ruminants - Nitrogen

Dietary protein entering the rumen is degraded to amino acids and ultimately ammonia in an apparently uncontrolled way. The ammonia that is not used for the synthesis of microbial protein is excreted in urine. The main reason for the low efficiency of capture of dietary N (Table 4.1) is an imbalance in the supply of degraded N relative to available energy (ATP), resulting in substantial amounts of ammonia being absorbed from the rumen [22, 23]. As a result of the relatively poor utilization of forage N (in the case of leafy forages like grasses, legumes, *Brassica spp*.), it is necessary to provide additional protein (or additional digestible energy, like maize silage, in the case of ruminants on pasture or conserved grass) for high producing stock. Although additional protein may elicit a production response, it inevitably leads to a greater loss of N to the environment.

Many different rumen microbial species, employing a range of proteolytic enzymes, are involved in the degradation of dietary protein. As a result, manipulation of this process has proved practically impossible to achieve in any consistent way. A number of alternative approaches have been examined as a means of improving utilization of dietary protein and reducing N excretion:

- *Reduce the rate of N application to grass, avoiding excessive rates*. On intensive dairy farms in Western Europe, this is the most effective measure, because it only has a small effect on herbage yield, whereas it reduces production of urinary N and increases potential uptake of urinary N by the sward [24].
- *Development of supplements to reduce protein degradability.*
- *Increase capture of dietary N by rumen micro-organisms,* using high-energy low-protein supplements [25, 26].
- *Breeding crops for improved N utilization.* Plant breeders in the UK have recently developed a number of varieties of high-sugar ryegrasses [27]. Research has shown that the increased sugar levels in the grasses can lead to more efficient use of grass protein by livestock, and less N excretion, suggesting that these grasses have the potential to reduce the environmental impact of livestock on forage-based diets.
- *Use of alternative forages.* A number of studies have confirmed the benefits, in terms of increased productivity and reduced N excretion, of including alternative forages in grass silage-based diets (high-energy low protein crops; crops with low protein degradability, like *Lotus sp.*). Table 4.6 summarizes the results of a number of studies on grass, maize, cereal or legume silages.

TABLE 4.6. SUMMARY OF MEAN EFFICIENCY OF MICROBIAL PROTEIN SYNTHESIS (EMPS) VALUES FOR DIETS BASED ON VARIOUS SILAGE TYPES [28]

	Diets based on:			
	Maize silage	Grass silage	Cereal silage	Legume silage
Mean EMPS*	48.4	30.1	35.9	19.5
Standard deviation	19.71	7.20	4.21	3.44
No. of observations	86	17	9	6

* g microbial N per kg organic matter apparently digested in the rumen (OMADR)

These data support the view that maize silage-based diets support greater microbial protein synthesis than grass silage, and that this is probably due to energy (ATP) supply being higher as a result of the starch present in maize silage. The very low values in legume silages are probably related to a lower supply of fermentable energy.

4.3.4. Ruminants - Phosphorus

The results of many studies have confirmed that dietary P concentration is the dominating factor affecting faecal P excretion, and that dietary management should be taken as the first defense against P build-up on farms [29, 30].

While sheep and beef cattle may derive most of their P from forages, substantial quantities of purchased P are fed to dairy cows, either incorporated in compound feeds or as mineral supplements. In a recent UK study, feed P accounted for 65% of total P brought onto the farm[3]. Similar values have been reported in The Netherlands [29] and the USA [31]. These data confirm that dietary P management, particularly for dairy cows, may play a key role in reducing P imports on dairy farms [29, 32, 33].

Phosphorus supplementation of ruminant diets has been regarded as essential for profitable and sustainable livestock production. However, there is evidence that the levels of dietary P in dairy cow diets are significantly higher than recent research would suggest necessary [34–36]. Reasons for this have been reviewed recently [37] and include concerns over the variation in P contents of feeds and uncertainties associated with P availability in mixed diets.

Recent research in the USA [30, 35, 38] has confirmed that increasing dietary P concentrations resulted in higher concentrations of total P in the faeces and higher total P excretion. This is illustrated in Figure 4.1 for a dairy cow yielding 45 kg milk day^{-1} and consuming 24 kg DM day^{-1} of a ration with 66.7% digestibility [35], where increasing P intake is reflected in increasing P excretion. The range in P intake (60 to 132 g day^{-1}) is equivalent to dietary P concentration of between 2.5 and 5.5 g $(kg\ DM)^{-1}$. This latter figure is typical of many diets for dairy cows being fed in Europe today, while data from current research would suggest that dietary concentrations nearer 4.0 g $(kg\ DM)^{-1}$ may be adequate [39, 40]. For the levels of DM and P intake mentioned, a reduction in dietary P content to 4.0 g kg^{-1} would result in a 38% reduction in P excretion. In The Netherlands, dietary P concentrations of 3.2–3.9 g $(kg\ DM)^{-1}$ are considered adequate for productive dairy cows and in studies of [34] 2.8 g P $(kg\ DM)^{-1}$ was sufficient to meet the P requirement of dairy cows producing approximately 9000 kg of milk per lactation.

[3] NT2402: Impact of nutrition and management on N and P excretion by dairy cows. Defra, UK.

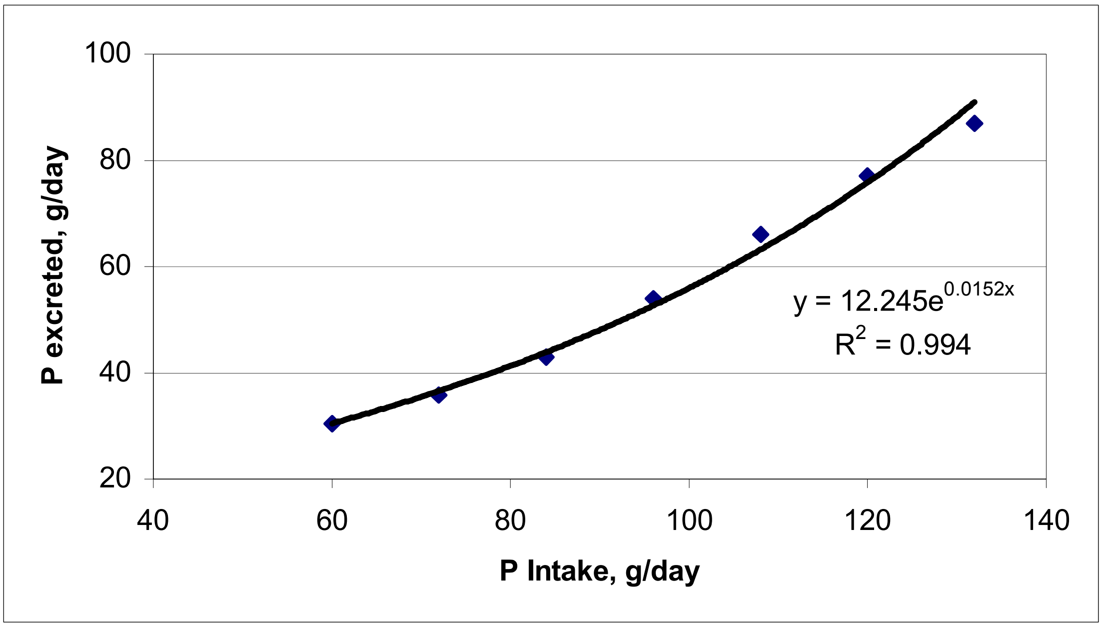

FIG. 4.1. Estimation of the P excretion of a lactating dairy cow yielding 45 kg milk day^{-1} and consuming 24 kg DM day^{-1} of a ration with 66.7% digestibility [41].

In a recent whole-farm study in the US [42], precision P feeding reduced P intake in 2 herds from 153% to 111% of NRC estimates of requirements [39]. This resulted in an estimated reduction in P excretion of 11.8 kg cow^{-1} year^{-1} (33% of P excretion before implementation of precision feeding). These results were achieved without any apparent adverse effects on feed intake, milk yield or dairy cow fertility. Similarly, dietary P content at the De Marke experimental dairy farm in The Netherlands averaged *ca*. 3.5 g (kg DM)$^{-1}$ for many years without negative effects on animal performance [43]. The P surplus on this farm averaged 0.2 kg P ha^{-1} year^{-1} in the period 1993–2000 [44].

In a recent survey of dairy farmers in the USA, 84% reported that ration formulation was provided by professionals rather than the producers themselves [45]. Most producers were feeding more P than cows needed because it was recommended in the rations by these consultants. If P levels in the diets of dairy cows are to be reduced, it is the consultants and feed formulators, rather than dairy farmers, who need to be persuaded of the benefits of doing so, and to have confidence in data for dietary P supply.

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ANNEX TABLE A1. CALCULATION OF NITROGEN (N) PRODUCTION IN THE MANURE OF DAIRY COWS (AFTER [3])

Parameter	[2]	Dimension	Comments
General data			
Milk production	5000	kg cow^{-1} $year^{-1}$	fat-corrected milk
Cow weight	425	kg	live weight
Calf weight	25	kg	live weight
Calf production	0.60	-	per cow per year
Nitrogen intake			
Feed intake cow [1]	4278	kg cow^{-1} $year^{-1}$	dry matter
N content of diet	2.80	%	of dry matter
N intake cow	120	kg cow^{-1} $year^{-1}$	
Retention of nitrogen			
Liveweight gain cow	25	kg cow^{-1} $year^{-1}$	
N content gain cow	2.50	%	of live weight
N content calf	2.90	%	of live weight
N content milk	0.54	%	of fresh weight
N retention in cow gain	0.63	kg cow^{-1} $year^{-1}$	
N retention in calf	0.44	kg cow^{-1} $year^{-1}$	
N retention in milk	27.0	kg cow^{-1} $year^{-1}$	
Total N retention	28.1	kg cow^{-1} $year^{-1}$	
Nitrogen excretion			
per cow	92	kg cow^{-1} $year^{-1}$	
Nitrogen losses from building and manure storage			
total N loss	10	%	of N excreted
per cow	9	kg cow^{-1} $year^{-1}$	
Nitrogen in manure			
per cow	83	kg cow^{-1} $year^{-1}$	

[1] 52 g DM per kg metabolic bodyweight ($W^{0.75}$) per day + 0.5 kg per kg fat-corrected milk (FCM) [3]; [2] Shaded cells are those requiring input data for the livestock type and production system; other cells contain default data (which can be edited if specific information is available) or output data

ANNEX TABLE A2. CALCULATION OF PHOSPHORUS (P) PRODUCTION IN THE MANURE OF DAIRY COWS (AFTER [7])

Parameter	[2]	**Dimension**	**Comments**
General data			
Milk production	5000	kg cow^{-1} $year^{-1}$	fat-corrected milk
Cow weight	425	kg	live weight
Calf weight	25	kg	live weight
Calf production	0.60	-	per cow per year
Phosphorus intake			
Feed intake cow [1]	4278	kg cow^{-1} $year^{-1}$	dry matter
P content of diet	0.44	%	of dry matter
P intake cow	18.8	kg cow^{-1} $year^{-1}$	
Retention of nitrogen			
Liveweight gain cow	25	kg cow^{-1} $year^{-1}$	
P content gain cow	0.75	%	of live weight
P content calf	0.60	%	of live weight
P content milk	0.105	%	of fresh weight
P retention in cow gain	0.19	kg cow^{-1} $year^{-1}$	
P retention in calf	0.09	kg cow^{-1} $year^{-1}$	
P retention in milk	5.25	kg cow^{-1} $year^{-1}$	
Total P retention	5.5	kg cow^{-1} $year^{-1}$	
Phosphorus excretion			
per cow	13.3	kg cow^{-1} $year^{-1}$	
per litre FCM	2.7	g $(kg\ FCM)^{-1}$	

[1] 52 g DM per kg metabolic bodyweight ($W^{0.75}$) per day + 0.5 kg per kg fat-corrected milk (FCM) [3]; [2] Shaded cells are those requiring input data for the livestock type and production system; other cells contain default data (which can be edited if specific information is available) or output data

ANNEX TABLE A3. CALCULATION OF N PRODUCTION IN THE MANURE OF SOWS WITH PIGLETS (AFTER [3])

Parameter	[1]	**Dimension**	**Comments**
General data			
Live piglet production	20	-	per sow per year
Weaning weight piglets	7.5	kg	live weight
Final weight piglets	25	kg	live weight
Nitrogen intake			
Feed conversion piglets	1.80	kg kg^{-1}	kg feed per kg gain
Feed intake sow	1140	kg sow^{-1} $year^{-1}$	dry matter
Feed intake piglets	630	kg $year^{-1}$	dry matter
N content of sow feed	2.60	%	of dry matter
N content of piglet feed	3.00	%	of dry matter
N intake sow	29.6	kg $year^{-1}$	
N intake piglets	18.9	kg $year^{-1}$	
Total N intake	48.5	kg $year^{-1}$	per sow + piglets
Retention of nitrogen			
Liveweight gain sow	40	kg $year^{-1}$	
Liveweight production piglets	500	kg $year^{-1}$	per sow
N content gain sow	2.50	%	of live weight
N content gain piglet	2.50	%	of live weight
N retention in sow	1.0	kg $year^{-1}$	
N retention in piglets	12.5	kg $year^{-1}$	
Total N retention	13.5	kg $year^{-1}$	per sow-place
Nitrogen excretion			
per sow	28.6	kg $year^{-1}$	
piglets	6.4	kg $year^{-1}$	
per sow + piglets	35.0	kg $year^{-1}$	per sow-place
Nitrogen losses in building and manure storage			
N loss building	21	%	of N excreted
N loss manure storage	5	%	of N stored
total N loss	25.0	%	of N excreted
per sow	7.2	kg N $year^{-1}$	
piglets	1.6	kg N $year^{-1}$	
per sow + piglets	8.8	kg N $year^{-1}$	per sow-place
Nitrogen in manure			
per sow	21.5	kg N $year^{-1}$	
piglets	4.8	kg N $year^{-1}$	
per sow + piglets	26.3	kg N $year^{-1}$	per sow-place

[1] Shaded cells are those requiring input data for the livestock type and production system; other cells contain default data (which can be edited if specific information is available) or output data

ANNEX TABLE A4. CALCULATION OF N PRODUCTION IN THE MANURE OF GROWING-FINISHING PIGS (AFTER [3])

Parameter	[1]	Dimension	Comments
General data			
Production cycle	115	Days	
Empty period	7	Days	between cycles
Starting weight	25	Kg	live weight
Slaughter weight	105	Kg	Live Weight
Rounds per year	3.0	-	
Nitrogen intake			
Feed conversion	2.90	kg kg^{-1}	kg feed per kg gain
Total feed intake	232	kg $animal^{-1}$	dry matter
Feed intake per phase			
Phase 1	232	kg $animal^{-1}$	dry matter
Phase 2	0	kg $animal^{-1}$	dry matter
N content of feed			
Phase 1	2.80	%	of dry matter
Phase 2		%	of dry matter
N intake			
Phase 1	6.50	kg $animal^{-1}$	
Phase 2	0.00	kg $animal^{-1}$	
Total N intake	6.50	kg $animal^{-1}$	per cycle
Retention of nitrogen			
Liveweight production	80	kg $animal^{-1}$	
N content	2.50	%	of live weight
N retention	2.00	kg $animal^{-1}$	per cycle
Nitrogen excretion			
Per animal	4.5	kg $animal^{-1}$	per cycle
Per animal place	13.5	kg $(animal\ place)^{-1}$ $year^{-1}$	3 rounds per year
Nitrogen losses in building and manure storage			
N loss building	21	%	of N excreted
N loss manure storage	5	%	of N stored
Total N loss	25.0	%	of N excreted
Per animal	1.1	kg $animal^{-1}$	per cycle
Per animal place	3.4	kg $(animal\ place)^{-1}$ $year^{-1}$	3 rounds per year
Nitrogen in manure			
Per animal	3.4	kg $animal^{-1}$	per cycle
Per animal place	10.1	kg $(animal\ place)^{-1}$ $year^{-1}$	3 rounds per year

[1] Shaded cells are those requiring input data for the livestock type and production system; other cells contain default data (which can be edited if specific information is available) or output data

ANNEX TABLE A5. CALCULATION OF N PRODUCTION IN THE MANURE OF LAYING HENS (AFTER [3])

Parameter	[1]	Dimension	Comments
General data			
Production cycle	405	days	
Empty period	14	days	between cycles
Occupancy	97	%	of time
Starting weight	1300	kg	live weight
Slaughter weight	1900	kg	live weight
Nitrogen intake			
Feed conversion	2.50	kg kg^{-1}	kg feed per kg eggs
Feed intake	45	kg $animal^{-1}$ $cycle^{-1}$	dry matter
N content of feed	2.80	%	of dry matter
N intake	1.26	kg $animal^{-1}$ $cycle^{-1}$	
Retention of nitrogen			
Liveweight gain	0.600	kg $animal^{-1}$ $cycle^{-1}$	
N content gain	2.80	%	of live weight
Egg production	18.00	kg $animal^{-1}$ $cycle^{-1}$	
N content eggs	1.85	%	
N retention in gain	0.02	kg $animal^{-1}$ $cycle^{-1}$	
N retention in eggs	0.33	kg $animal^{-1}$ $cycle^{-1}$	
Total N retention	0.35	kg $animal^{-1}$ $cycle^{-1}$	
Nitrogen excretion			
per animal	0.91	kg $animal^{-1}$ $cycle^{-1}$	
per animal place	0.79	kg (animal place)$^{-1}$ $year^{-1}$	
Nitrogen losses in building and manure storage			
N loss building		%	of N excreted
N loss manure storage		%	of N stored
Total N loss	30	%	of N excreted
per animal place	0.24	kg (animal place)$^{-1}$ $year^{-1}$	
Nitrogen in manure			
per animal place	0.56	kg (animal place)$^{-1}$ $year^{-1}$	

[1] Shaded cells are those requiring input data for the livestock type and production system; other cells contain default data (which can be edited if specific information is available) or output data

5. MANURE MANAGEMENT DURING HOUSING AND STORAGE

Use of animal manures for plant production is probably the most cost-efficient alternative to direct discharge to surface waters. Managed properly, the recycling of the effluent will represent a low risk of disease transmission and the benefits of recycling, *viz*. reduction in the use of chemical fertilizers and better soil quality, may prove economically attractive to the farmer. An important aspect of sustainable manure management is to develop housing and manure storage systems that help to conserve the plant nutrients and maintain a high concentration of plant nutrients in the manure. The latter requires limitation of water use. This will help to defer the costs of management, including transport.

5.1. HOUSING

Excreta from animal houses can be handled as liquid or solid manure. In Asia, in-house separation of the excreta produces liquid and solid manure from buildings with either slatted floors or solid concrete floors (Chapter 2). The solid fraction is collected manually by scraping the slats or the solid floor below the slats. The floor is often further cleaned by hosing after the solids have been removed. The liquid drains through channels to lagoons and rivers. In some animal houses, the manure may be removed mainly by hosing and a slurry is produced containing faeces, urine, water and some feed residues and bedding material. Most studies have proven that, in the tropical or subtropical parts of Asia, deep litter systems for pig production are not feasible due to the heat production in the litter by microbial fermentation. Generally, the animals require cooling, either by hosing, by misting the animals with a fine spray, or by spraying the roofs with sprinklers.

The amount of nutrients in the solid fraction of pig manure may be estimated from Dutch experiments with in-house separation of excreta (Table 5.1; [1]). In these experiments, about 35% of the excreta were collected in the solid fraction, which had a high concentration of dry matter and nutrients. In a feeding experiment in The Netherlands [2], fattening pigs (body weight 55–106 kg) produced, on average, *ca*. 930 g faeces and 3,500 g urine per day with, respectively, 0.92% and 0.46–0.81% total N. The variation in urinary N content was caused by the N content of the diet. Compared to these urinary N contents, total N in the liquid fraction in Table 5.1 was very low, indicating addition of spilt drinking water or cleaning water.

TABLE 5.1. AVERAGE COMPOSITION OF THE LIQUID AND SOLID FRACTION AFTER IN-HOUSE SEPARATION ON FILTER-NETS BELOW A PARTIALLY SLATTED FLOOR IN PIGGERIES [1]

Fraction	DM	Ash	N-total	NH_4^+-N	P_2O_5	K_2O	CaO	MgO	Cu (ppm)	pH
	(% of fresh weight)									
Liquid	1.92	1.21	0.34	0.35	0.05	0.62	0.04	0.02	2.5	9.1
Solids	32.5	8.34	1.24	0.34	1.64	0.85	1.45	0.48	189	-

The composition of liquid manure on commercial pig farms and the costs of manure storage and transport will be influenced directly by the amount of water from cleaning the floors, cooling the animals, rain and surface run-off to the lagoons. The volume of liquid manure or slurry, therefore, to a great extent depends on the amount of water used for cleaning and cooling. In the Singapore study ‘Pig Waste Management and Recycling’ [3], the amount of liquid manure (faeces + urine + spilt drinking water) produced by pigs was estimated at *ca*. 4.5 kg per day per standing pig population unit (SPP)[4]. When cleaning the floors with hoses, the amount of water used was 20 litres per SPP (7.3 m^3 SPP^{-1} $year^{-1}$) and, in a flushing system, it was 30 litres per SPP. If the pigs are cooled by hosing, an additional 5–10 litres is used per SPP.

In Asia, water use on pig farms is very high and should be reduced as much as possible if manure is to be recycled at low costs. Thus, cooling through sprinkling of the pigs or by fine atomized spray (mist) is more efficient than hosing, both in terms of cooling the animals and in water use. The sprinklers can be adjusted so that the added water is evaporated, which is very efficient for cooling the animals; thereby, little cooling water is added to the liquid manure. No water from cooling should be added to the liquid manure if the cooling water is sprinkled over the roof of the animal house or if the house is cooled by drawing air into the animal accommodation through a curtain of water in the gable end.

Water use may also be reduced by mitigating ineffective water use for cleaning and drinking water supply from leaking drinkers. By using water-saving techniques (*e.g.* high-pressure cleaning and low-pressure drinkers), the volume of effluent can be reduced. The volume of water needed for cleaning animal houses may also be reduced by pre-scraping floors with brooms or other scraping equipment and by flushing channels or pipes with the separated liquid following sedimentation or mechanical separation.

If the solids have been separated out, then the liquid manure may be more easily transferred by gravity to the store. In slurry systems, sedimentation of solid material may impede the transport of the slurry and, instead, the slurry should be pumped to the slurry store. To facilitate transport of slurry from the animal house to the main store, a small intermediate store may be constructed near the house. From the animal house the liquid will run to the intermediate store, the slurry should be transferred by a push and plug system ensuring that solids do not sediment in the channels within the animal house. Once or twice a week, the intermediate store is then emptied by pumping the liquid to the lagoons, the intermediate store acting as a pump sump. The pipes used for transportation should be PVC pipes that may be buried below ground.

In areas of high rainfall, the roofs should overhang the side of the building by about 1 meter and rain gutters will ensure that the effluent is not diluted with the clean roof water. Rain water should thus be kept separate from the manure or effluent and, along with rain collected on clean concrete surfaces, should be transported separately to a suitable reservoir for receiving water, which may be a river or a lake. The rationale is that rain will increase greatly the volume of the liquid manure, which will also increase the need for storage capacity and the volume of liquid that has to be transported to the field. Consequently, the costs of recycling will increase considerably.

The efficiency of the techniques for reducing water use may be estimated by evaluating the concentration of total suspended solids, phosphorus or total-N in the effluent. On a pig farm with efficient water use (only 20 litres of water per SPP for cooling and

[4] SPP is an estimate of the number of pigs (sows, piglets, weaners, porkers and boars), and is calculated by multiplying the number of pen spaces assigned to sows or to sows + gilts by a factor between 10.5 and 12 [3]. Average weight of an SPP unit in the Singapore study was 54 kg. There is a need for a more functional and management-related standard than SPP (*e.g.* well defined animal categories, like sows + piglets, weaners, porkers), because information expressed per SPP cannot be used in different pig production systems.

cleaning), the concentrations of these components should be as follows: total-P ≅ 0.4 g $litre^{-1}$, total-N ≅ 1.2 g $litre^{-1}$ and total solids (dry matter) ≅ 18 g $litre^{-1}$.

5.2. LIQUID MANURE STORES FOR PIG EFFLUENT

Storage of pig effluent will be required in order to gain the maximum benefit of the effluent for fertilizing crops, grassland and plantations, because good timing of application in relation to the crop's nutrient requirement is important (Chapter 7). Storage may also be needed as a means for quality control of the effluent, *e.g.* allowing measurement of plant nutrients and control/reduction of pathogens.

Lagoons for holding livestock effluents can be constructed cheaply, but they should be lined for groundwater protection purposes. Leakage or leaching from un-lined lagoons may be variable, because this is related to the soil type and the nature of the slurry. The lagoons should preferably have a sealed bottom and sides. Lagoons excavated in soil with a high clay content may not need lining, due to the very low liquid infiltration rate, particularly where the base is compacted during construction. On other soil types, lining with PVC-membranes or construction with concrete will stop leaching, whereas the liquid level in the lagoon may be managed by recycling the effluent to crops. Effluent lagoons should be bordered by embankments that exclude addition of surface run-off water. Furthermore, in high rainfall areas, covers to reduce the addition of rainwater to the lagoon would be a worthwhile investment, because lagoons are usually shallow and do not normally have spare capacity for additional rainwater. If the net precipitation (rain minus evaporation) is 600 mm, then 60 cm of the lagoon's depth would be occupied by rain water, as a result of rain water incident over the surface area of the lagoon, only.

An alternative to lagoons may be under-floor effluent stores or concrete pits outside the building. Liquid manure stores inside the buildings will be protected against the addition of rain water. Concrete pits, which are partly below ground level may prove cheaper to construct than tanks buried in the ground, but slurry would need to be pumped to above-ground stores.

Solids in the effluent may settle in the base of the stores or accumulate at the surface, producing a crust. In stored pig slurry the solids settle, whereas in cattle slurry both settling and crust formation are observed. If the slurry is not mixed thoroughly before and during emptying the store, then the store has to be de-sludged periodically. Otherwise, the storage capacity may diminish considerably with time. Due to sedimentation and microbial transformation of the organic content of effluent, storage may reduce considerably the solids and nutrient content of the effluent (Table 5.2).

TABLE 5.2. EFFECTIVITY OF LAGOONS FOR REDUCING NUTRIENTS AND POLLUTANTS IN PIG EFFLUENT FOLLOWING PASSAGE THROUGH TWO OR MORE LAGOONS IN SERIES (SEE FIGURE 2.3)

Source	Fraction removed (% of the amount at the start of storage)				
	P	NH_4^+-N	Soluble COD[1]	Total solids	Volatile solids
Burton, 1997 [4]	50–90	60	70		
Taiganides, 1992 [3]			45	42	33

[1] COD = Chemical Oxygen Demand

5.3. MITIGATION OF PATHOGENS IN ANIMAL EFFLUENT

Solid manure and animal liquid effluent may contain pathogens that are a significant health hazard to both humans and animals. The pathogens that may be transferred with liquid manure belong to groups like bacteria, viruses, protozoa and parasites; some examples are presented in Table 5.3. Before recycling manure for plant production, it is important to know if the herd is infected with high-risk pathogens and to take account of the risk of transmission in manure. Manure may be treated for the purpose of ensuring that pathogens are not spread.

TABLE 5.3. EXAMPLES OF PATHOGENS THAT MAY BE SPREAD WITH LIQUID EFFLUENT AND SOLID MANURE

Class/phylum	Species	Disease
Bacteria	*Salmonella* spp.	Gastro-enteritis (‘stomach flu’)
	Treponema (Brachyspira) hyodysenteriae	Swine dysentery
	Faecal streptococci	Gastro-enteritis (‘stomach flu’)
Viruses	Herpes virus	Aujezsky’s disease
	Aphtho virus	Foot and mouth disease
Protozoa	*Giardia* spp.	Giardiasis (beaver fever)
Nematoda (worms)	*Ascaris* spp.	Ascariasis (parasitic roundworm)
Trematoda (worms)	*Schistosoma* spp.	Schistosomiasis (bilharzia)
Cestoda (worms)	*Taenia saginata*	Taeniasis (tapeworm)

Discharging pig slurry from infected premises to rivers represents a great risk for the spreading of the pathogens. There is strong evidence that spreading of livestock slurries on agricultural land provides less risk for transmission of pathogens to humans and animals than direct discharge to rivers that may subsequently be used for water abstraction for domestic use, stock water supply, and irrigation of crops and pastures.

The numbers of bacteria and viruses in animal effluents are often recorded on a logarithmic scale, and the number of *Escherichia coli* in pig slurry may be about $10^{5.5}$ units and faecal *streptococci* $10^{4.3}$ units (10^3 to $10^{5.7}$) per g of biomass. The numbers of *Salmonella* are not counted in thousands, nor are the number of protozoa or parasites.

To reduce pathogens, animal manure may be treated aerobically *via* composting or in anaerobic or aerated lagoons. Temperature and treatment time are the two most important process parameters, when evaluating process variables for the reduction of pathogens in solid manure and liquid effluent [5]. During composting of solid manure, aerobic decomposition of organic matter and impeded heat transport cause heating of the material, often to *ca*. 60–70^{o}C [6]. This heating has positive effects by killing pathogens and weed seeds. At low temperatures, the reduction rate of pathogens is slow, thus, after a lagoon treatment period of more than 120 days, the concentrations of micro-organisms remaining in the effluent from lagoons in Europe were high, *viz*. 10^5 per 100 ml for faecal coliforms and faecal *streptococci* and 10^4 per 100 ml for *Clostridia* [4]. Due to higher ambient temperatures, storage of liquid effluents may be a more efficient and reliable treatment in Asia than in Europe, but the efficiency of storage on pathogen reduction should be assessed before using storage as the sole treatment measure.

The continual addition of effluent to a lagoon will affect the effective retention time. Fresh additions of slurry may short-circuit the nominal retention time to much less than the hydraulic retention time, thereby greatly reducing treatment effectivity. Therefore, the storage and treatment of the liquid effluent may be more effective as a batch operation, which will ensure that the real retention time and treatment retention time are similar.

Composting may facilitate production of hygienic solid manure that may be applied to land with minimal risk from pathogens. Sedimented or separated solids and poultry manure can be treated this way, as well as slurry mixed with straw or other porous organic solid residues. During composting, the temperature of the material should exceed 55–65^{o}C for at least one week to give a good reduction in pathogens and weed seeds [5–8]. These conditions are generally achieved during normal composting. Heating of the material may be favoured by storing the solid organic residue in windrows and turning the windrows every 7 days for some weeks or, in case of rather dry organic residues (> 70% dry matter), by adding water.

5.3.1. Regulations of effluent discharge

Livestock slurry may be used for plant production without any restrictions in most countries. Thus, for the purpose of evaluating spreading of pathogens and treatment options of animal slurry, one may use the rules for sewage sludge, industrial wastes, *etc.* [9]. These regulations may also be used to evaluate the risks associated with the existing discharge of pig slurry to surface waters and to define precautions that farmers should take before starting to use pig slurry for crop production.

Treatment will often reduce but not eliminate the content of pathogens. Due to the vast number of potential pathogens, a few test organisms have been used to evaluate the efficiency of a treatment, *viz*. *Escherichia coli* and faecal *streptococci*. Faecal *streptococci* are often recommended to be used for this purpose, because they are resistant to effluent treatment, in particular to heating. Furthermore, test viruses in special carriers have been used to evaluate virus reduction during fermentation. This technique ensures that no added viruses are spread in the environment [10].

Effluent treatment, in general, has to fulfill all or some of the following conditions:

(1) Process standards, *i.e*. the temperature and duration of the treatment.
(2) Use of specific test organisms in the treated effluent.
(3) Reduction in test organisms during treatment.

The efficiency of a treatment may be evaluated using the instructions of the German Veterinary Medical Association (DVG) for testing chemical disinfectants. They claim that a 4 log units reduction in the test germs is sufficient [11]. The test germs, used for evaluating the inactivation potential of a process could be *Escherichia coli* and *faecal streptococci*. In addition to this test, elimination of *Listeria*, *Yersinia* and *Salmonella* has been used to evaluate the efficiency of slurry sanitation in biogas plants in Danish studies [12]. US and WHO standards also include figures for the number of viable nematode eggs and cysts of Protozoa [13].

A study of the capacity of anaerobic fermentation reactors in Denmark to reduce bacterial pathogens concluded that *Escherichia coli* and faecal enterococci are suitable indicator organisms for this capacity [14]. Based on this study, the authors recommended that faecal enterococci should be adopted to give the broadest safety margin, and that an acceptable reduction capacity is 3–4 log units, similar to the German recommendations. The treatment should reduce the indicator bacteria to ≤ 102 units per ml.

In conclusion, use of faecal *streptococci* and *Escherichia coli* may be recommended in general for monitoring reduction of pathogens in animal effluent. In case of concern about the hygienic standards, other indicator organisms may be used that are related to the specific pathogens of concern. For this purpose, use of the following organisms may be recommended: bacterial spores – *Clostridia* spores; virus – parvovirus; bacteriophages – *Salmonella*; and parasites – *Ascaris suum* eggs [15].

5.4. MIXING AND PUMPING

In stored livestock slurry, suspended matter of higher density accumulates as a sludge on the bottom of the lagoon and, in cattle slurry, a crust may form. Due to sedimentation, the storage capacity of the lagoon will be reduced and the sediment should be removed periodically. Since it is difficult and costly to remove the sludge, it may be advantageous to separate the solids from the liquid manure before storage in order to reduce sedimentation. Alternatively, the liquid manure should be mixed before emptying the store. Mixing the slurry will also improve the homogeneity in nutrient composition and the accuracy of field application of slurry nutrients [16]. Furthermore, mixing will improve the rheological properties enabling a more constant flow during spreading operations. Mixing may release hazardous gases (H_2S or CH_4, which are, respectively, highly toxic and potentially explosive), but this is not a big problem when mixing lagoons, because they are not enclosed in a building.

The effluent may be mixed mechanically with rotating impellers. The impellers may be fixed to the wall of the store or lagoon, and are often powered by electric motors. Alternatively there are tractor-driven mixers that can be lowered into the lagoon using hydraulics. These are quite efficient because the mixer can be moved around the lagoon. One may also use hydraulic or jet mixing with pumps set within, or external to, the slurry store. Using hydraulic mixing, the pipes should have a diameter of 125 mm or more and, ideally, the system should include adjustable nozzles at the end of the return pipe.

Pumping will be needed when transferring slurry around the farm, *e.g.* between the livestock building and the slurry store. Furthermore, effluent may be pumped from the lagoon to the field, during application. The pumping distance is an important consideration, and effluent with a low solids content can be pumped for longer distances and at lower cost than effluent with a high solids content. Increasing dilution reduces slurry viscosity and problems associated with pumping but increases effluent volume and, therefore, the costs of storage and transport. A compromise is required between minimizing the volume of effluent that must be stored and handled and increasing the volume for ease of pumping. Alternatively, digestion (aerobic or anaerobic) or separation may reduce the solids content and the viscosity of effluents, thereby improving the flow properties. Liquid manure from animal houses with in-house separation may be transported by gravity through channels.

5.5. AMMONIA EMISSION

Nitrogen losses by NH_3 volatilization from livestock buildings and stored manure reduce the fertilizer value of livestock manures. In addition, this NH_3 increases the level of atmospheric N deposition and contributes in this way to eutrophication of natural ecosystems and associated loss of biodiversity, increased susceptibility of trees to stress, soil acidification and, in some cases, even too much nitrate leaching from affected ecosystems [17–20]. Furthermore, NH_3 emissions play a role in the formation of $PM_{2.5}$ and PM_{10} [5], airborne

[5] $PM_{2.5}$ and PM_{10}, particulate matter < 2.5 and 10 micrometers, respectively

particulates that can be a health hazard [21, 22]. Consequently, ceilings on the annual NH_3 emissions have been included in the Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (Gothenburg Protocol, agreed in 1999; [23]), and in the EC Directive on National Emission Ceilings for certain Atmospheric Pollutants (Directive 2001/81/EC; [24]). Another point of concern related to NH_3 emission is that high NH_3 concentrations in livestock buildings adversely affect human and animal health [25].

Ammonia volatilization takes place when animal manure is exposed to the air. The main factors affecting NH_3 volatilization from manure are the content of total ammoniacal nitrogen (TAN = NH_4^+ + NH_3), emitting surface area, animal behaviour, manure pH and temperature, and weather conditions [26–29]. Weather factors stimulating water evaporation, like temperature, radiation, relative humidity, and wind speed generally stimulate NH_3 volatilization. Although faeces and urine of mammals hardly contain TAN at the moment of excretion, NH_4^+ is formed rapidly by hydrolysis of urea, the main N compound in urine [26]. This process leads to an increase in pH and, as a consequence, to the formation of dissolved NH_3 and, hence, to NH_3 volatilization. Urea hydrolysis is catalyzed by the enzyme urease, produced by micro-organisms in faeces. These micro-organisms are also present in a layer of precipitated minerals on fouled floors of livestock buildings [30] and in soils. Although urea is the main source of TAN in ruminant and pig manure and of NH_3 volatilization, the urinary components allantoin and creatinine also contribute to it, whereas hippuric acid, another urinary N compound, stimulates hydrolysis of urea and subsequent NH_3 loss [31]. Urea is converted to TAN within *ca*. 2 hours after urine deposition on a fouled floor at temperatures of 10^{o}C or higher [32]. The omni-presence of urease-producing bacteria makes it difficult and costly to inhibit this process. On the contrary, reducing dietary N content, without compromising protein supply to the animal, is a convenient method to reduce N excretion in urine and NH_3 emission [2, 33–35].

The main N compound in poultry excreta is uric acid. Generally, total N in droppings (fresh excreta) of laying hens consists of 60–75% uric acid, 0–3% urea, 0–3% NH_4^+-N, and 25–34% undigested proteins [36]. The enzyme uricase is specific for the degradation of uric acid. It is an endogenous enzyme, generally present in micro-organisms. Degradation of uric acid to NH_4^+-N is influenced by temperature, pH, moisture content, and oxygen supply [36]. The degradation is strongly limited at manure temperatures < 20^{o}C, pH values < 6.0, moisture contents < 40%, and lack of oxygen. In general, degradation of uric acid requires much more time than hydrolysis of urea. Decomposition rates of 8 and 40% of the amount of uric acid per day have been reported for dry and liquid poultry manure, respectively [37]. However, information on this aspect is limited. As a consequence of the slow degradation of uric acid and the factors involved in this process, the concentration of TAN and uric acid in poultry manure is variable and prediction of NH_3 emission should include degradation of uric acid. Rapid drying of poultry excreta is a method employed widely to reduce NH_3 emission from poultry buildings and stored poultry manure [36].

In future, it is hoped that more information about N transformations in manure managed under Asian conditions becomes available. This information may be useful in calculations that will help to account for changes in manure N due to gaseous N emissions (NH_3, and to a lesser extent NO, N_2O and N_2), and mineralization or immobilization of N during manure storage [38].

Most important processes affecting NH_3 emission from cattle and pig buildings and manure stores have been reviewed in [29]. This includes NH_3 emission factors[6] for different types of cattle and pig buildings and manure stores as well as emission reduction measures. Information on relevant processes, NH_3 emission factors and reduction measures in poultry housing systems is available in [36, 39].

5.5.1. Housing

Ammonia emission factors for commercial pig houses with fully slatted floors in Western Europe vary between 17 and 29% of excreted N [29, 40, 41]. In many Asian countries, average outdoor temperature is *ca*. 10–15°C higher than in Western Europe, and studies have indicated that NH_3 volatilization is stimulated by higher temperatures [26]. On the other hand, frequent flushing of the floor with water may reduce emission by as much as 50% [42], and in most Asian pig production systems the floor is flushed frequently. These opposite effects and other factors affecting NH_3 emission hamper estimates of NH_3 emission from livestock buildings in Asia. Local measurements are necessary to make these estimates and to develop appropriate emission reduction technology.

5.5.2. Manure stores

Ammonia volatilization factors for manure stores or lagoons in pig and cattle production systems are given in Table 5.4. In Asia, there will be no crust of organic material on the stored effluent as no bedding materials are used in the pens. The higher temperature in Asia will increase the NH_3 volatilization potential but, conversely, a lower concentration of NH_4^+-N in the slurry because of liberal water use will reduce the potential; therefore, it is assumed that the NH_3 volatilization from the liquid manure is only slightly higher than the figures presented in Table 5.4. Ammonia volatilization from the solids is considered to be similar to the emission measured in Europe, because volatilization is related to the temperature increase due to composting of the solids.

TABLE 5.4. AMMONIA EMISSION FACTORS FOR STORED MANURE IN WESTERN EUROPE AND NORTH AMERICA (SOURCE: [29])

Manure and storage type	Emission factor (kg NH_3-N m^{-2} $year^{-1}$)	
	Mean	SD
Pig slurry without surface cover (lagoons)	0.78	1.07
Pig slurry without surface cover (concrete store)	2.18	2.10
Cattle slurry without surface cover (concrete store)	1.44	0.78
Cattle and pig slurry from biogas plant (concrete store)	2.33	0.68
Solid manure (composting)	15–30% of total N ex animal house	

5.6. CONCLUSIONS

Direct discharge of liquid manure to surface waters is a great risk to human and animal health and causes serious eutrophication of rivers, lakes and coastal waters. Recycling of manures for plant production is probably the most cost-efficient alternative. At present, many livestock farms in Asia produce excessive volumes of liquid manure due to liberal use

[6] NH_3 emission factors are estimates of average NH_3 emission from livestock buildings or manure stores. They are expressed as annual emission per animal or animal place, as proportion of total N excreted or stored, or as annual emission per square meter fouled floor or slurry surface.

of water for cleaning the building and cooling the animals. This hampers proper use of the manure for plant production. Therefore, much attention should be given to the development of housing and manure management systems with minimal use of water.

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6. PROCESSING AND HANDLING OF MANURE TO REDUCE POLLUTION AND IMPROVE NUTRIENT UTILIZATION

6.1. INTRODUCTION

This chapter introduces and reviews some of the principles and methods of handling and processing of livestock manures and effluents. It comprises (i) separation or settling processes which aim at obtaining a solid phase (or a sludge) and a clarified liquid, (ii) biological processes (aerobic and anaerobic) devoted to the breakdown of organic compounds and nitrogen and carbon removal, (iii) combined processes involving physical, chemical and biological interactions. The main objectives for implementing manure processing systems are:

- easier handling,
- abatement of odours and water and air pollution,
- biogas production,
- production of a more valuable organic fertilizer.

Further to the basic principles and mechanisms of manure processing, the expected performances, experiences, and the suitability of implementing such processes in Asian livestock situations are discussed. Some case studies on manure processing are also presented.

The prevailing manure management systems in Western Europe are strongly affected and determined by conflicting requirements. These are embedded in the concept of sustainability. This concept embraces requirements from society (quality of life), economy (farm and rural income) and ecology (nature and environment). The words people, profit and planet are often used to refer to these aspects of sustainability. Some general characteristics of European manure management systems are:

- The type of manure on cattle and pig farms is predominantly slurry, a mixture of faeces and urine and often some water and feed residues. Most poultry farms produce dry manure with dry matter (DM) contents between 40 and 70% [1].
- Manure storage facilities are required with a capacity for several months to bridge the autumn and winter season when land application of manure is not recommended and often not allowed by legislation.
- Almost all livestock manure is used to fertilize crops and grasslands, generally without processing.
- Most countries enacted legislation to limit pollution of the environment by livestock manures [2]. This includes regulations related to (i) type and capacity of manure storage facilities, (ii) period, rate and method of manure application, and (iii) possibilities to extend the size of a livestock farm, particularly close to built-up areas and nature reserves. The European Union enforces this type of legislation.

Sustainable manure management implies effective use of manure nutrients in the fertilization of crops and grasslands and a corresponding reduction in the rate of application of inorganic fertilizers. A basic concept of such a strategy is, therefore, to move away from considering manures as waste, and promoting their value as organic fertilizers. However, where there is a local surplus of manure relative to available land for application, good management alone will not be enough to avoid pollution. Surpluses need to be transported out of the area and some form of manure processing may facilitate this.

The development of new manure treatment technologies and the best ways to apply them has been the subject of extensive research. A wide range of such technologies is available, which can contribute to mitigation of pollution. Some of these technologies are particularly recommended for reducing nutrient content (N, P) or biochemical oxygen demand (BOD). The objectives of the manure management for a farm should be clearly defined and the chosen treatment process should be an integral part of the manure management system and capable of meeting these objectives. It is essential that the objectives are identified and made explicit before considering any treatment option [3].

Advanced technologies are available to convert livestock slurry into clean water that may be discharged into watercourses or even be used as drinking water. However, such technologies require heavy investments and are very costly. With respect to manure handling and processing, there is a strong need to identify and develop practical systems that solve local problems at minimum costs. This requires a good definition of the problems to be addressed and identification/assessment of the practical options to solve them. So, partial treatment may be appropriate and inexpensive. 'Treatments' can also be considered to include a number of 'passive' options (*e.g.* settlement, storage, passive separation of solids *via* strainer facilities during storage, dilution), as well as 'active' options more typically regarded as treatment. In some cases, a financial value can be ascribed to this, such as energy savings from the use of generated biogas, sales of organic by-products or reduced purchases of inorganic fertilizers. The more important and relevant manure processing options are considered within this chapter.

6.2. PHYSICAL PROCESSES (SOLIDS-LIQUID SEPARATION)

6.2.1. General principles

The relatively simple process of separating manure solids and liquid can offer advantages in terms of improved handling and management characteristics of the two products. In addition, it allows application of the different manure components (organic matter, nutrients) on fields and crops where they will be most effective. Separation may be particularly useful in regions with a manure surplus, *i.e.* in regions where the content of nutrients in animal manures exceeds the requirement of the crops. It allows application of the liquid fraction to local crops, and transport of the solids containing a fraction of the nutrients to other farms or regions, sometimes after conditioning (drying, pelletizing, composting), at relatively low cost.

There are two basic methods of solids-liquid separation. One uses the difference in density between the solid particles and the liquid (sedimentation and centrifuging) and the other one uses the shape and size of the particles to cause separation (screening and filtration).

6.2.1.1. Sedimentation

The easiest way to remove suspended solid material from liquid manure is by utilizing natural settling or sedimentation [4]. The sedimentation option appears to be an attractive method for removing fine solids from slurry because of the relative simplicity of the process and the low costs of the equipment involved.

6.2.1.2. Mechanical separation

A quicker separation can be obtained using mechanical screening, a technique easily applicable on farms to separate the coarse solids from the slurry. Mechanical screening is also an initial process step in many complete treatment processes [5].

6.2.2. Performance and examples

Performance of mechanical separation systems is usually assessed in terms of slurry flow rate and relative output of solids and liquid, with separation percentages of solids and the macro-nutrients N, P and K. As an illustration, data from the more common separators are shown in Table 6.1.

TABLE 6.1. SEPARATION EFFICIENCY AND TECHNICAL DATA FOR COMMON SEPARATORS [3]

	Belt press	Sieve drum	Screw press	Sieve centrifuge	Decanter centrifuge
Flow rate, m^3 $hour^{-1}$	3.3	8–20	4–18	1.9–5.5	5–15
Separation efficiency, %					
DM	56	20–62	20–65	13–52	54–68
N	32	10–25	5–28	6–30	20–40
P	29	10–26	7–33	6–24	52–78
K	27	17	5–18	6–36	5–20
Volume reduction, %	29	10–25	5–25	7–26	13–29
Specific energy, kWh m^{-3}	0.7	1	0.5–2.0	2.2–6.7	2.0–5.3

The performance of most separators varies widely (Table 6.1). This is possibly caused by both the set-up of the equipment and the characteristics of the slurry. In most cases, only the separation efficiency for DM is significantly higher than volume reduction, whereas the separation efficiencies for the nutrients are in the same order of magnitude as volume reduction. This indicates hardly any concentration of nutrients in the solid fraction. Nevertheless, it has been stated that with suitable technology, *i.e.* correct equipment selection and set-up, a nutrient removal of up to 50% for P and 30% for N can be achieved. In this way, manure nutrients and organic matter can be concentrated in the solid fraction (only 10–20% of original mass) and may be transported at reduced costs to regions with organic matter and nutrient demand. However, the results in Table 6.1 indicate that this possibility should be considered with caution, because in most studies there was hardly any concentration of nutrients in the solid fraction.

After separation of solids and liquid, composting of the solids may be a next processing step. Generally, composting causes heating of the material by aerobic decomposition of organic matter and impeded heat transport, and heating stimulates evaporation of water. This further reduces the weight of the solid fraction and transportation costs. The liquid fraction, remaining after separation, may be irrigated to land close to the production unit or subjected to further treatment, prior to land application or discharge.

Shutt and coworkers [6] reported the removal of 35% solids, 62% BOD and 69% chemical oxygen demand (COD) from pig slurry, by a simple run-down screen, even with only 3% of the volume removed. Performance varied with the screen slot width (0.1 cm was better than 0.15 cm) and slurry inflow rate.

6.2.2.1. Membrane processes

Osmosis is based on the principle of a semi-permeable membrane, where water flows from the side with the lower salt concentration to the side with the higher concentration, thus creating the cell turgor (hydrostatic pressure). Reverse osmosis works with pressure on the side of the higher salt concentration, thus forcing the permeate (water) through the membrane, holding back the minerals and salts. Experiments carried out by [7], using a pilot plant (8 m^2

of membrane surface) under different conditions with settled sow slurry, have demonstrated a 99% salt retention. The extent of salt retention decreased as the membranes aged. The use of membrane technology in the dewatering of sow slurry is possible, but an important prerequisite is that the organic fraction should be decomposed and the solids removed by effective sedimentation, separation or filtering processes before the liquid enters any membrane treatment step.

6.2.3. Recommendations for these techniques

Separation equipment is usually reliable and robust. There is a wide range of devices from highly sophisticated to more simple ones. These include run-down screen, vibrating screen, belt press, drum press, press screw/auger separator, sieve centrifuge, decanter centrifuge. Costs vary widely reflecting sophistication and performance. At the low end are the basic screening packages, and at the high end the centrifuges.

The following advantages of slurry separation have been reported:

- increased slurry utilization, allowing the use of slurry components on crops and soils where they produce best effects and lowest pollution;
- improved homogeneity of the liquid phase, with less sedimentation and generally no crust formation;
- reduced slurry volume (and required storage capacity);
- easier handling of the liquid fraction, facilitating improved accuracy of spreading;
- reduced nutrients loading *via* slurry application (may be significant in cases of a surplus of manure nutrients);
- reduced energy requirement for mixing and pumping and reduced risk of blockages;
- improved infiltration of the liquid into the soil, for reduced odour and NH_3 emissions;
- reduced herbage contamination with slurry solids and, hence, reduced risk of negative impact on silage quality or pathogen transfer to grazing animals;
- useful pre-treatment for biological processing.

Some *disadvantages* also have to be considered:

- storage, handling and spreading of two separate materials;
- necessary investment in machinery;
- requirement of extra farm labour and technical skill.

6.3. BIOLOGICAL PROCESSES

6.3.1. Aerobic treatment

6.3.1.1. General principles

All organic matter, including that in animal manure, can be subjected to decomposition by aerobic and/or anaerobic micro-organisms. By optimizing the environment of the naturally occurring micro-organisms, it is possible to use these species for the specific purpose of biological treatment to produce useful end products. Aerobic treatment of livestock slurries is reported to cause the following effects:

- Biological stabilization of the slurry by decomposition of reactive organic matter (characterized by the BOD_5 content[7]).
- Removal of unpleasant odorous organic compounds, like volatile fatty acids, by their oxidation to produce odour-free substances such as carbon dioxide (CO_2) and water.
- Increased availability of some plant nutrients, and thus a higher value of the end-product as a fertilizer. This point needs to be questioned, because aeration indeed causes mineralization of nutrients contained in organic matter (*e.g.* N and P), but at the same time stimulates gaseous N losses. As a consequence, the content of inorganic N generally will decrease and it is questionable whether the mineralization of other nutrients will improve their availability.
- Reduction in pathogen content by inactivation of certain pathogens that are strictly anaerobic, such as Enterobacteriae (*Salmonella*, *Escherichia coli*), faecal *streptococci* and some viruses. Moreover, aeration of stored dry manures (farm-yard manure, naturally or artificially dried poultry manure, and the solid fraction after slurry separation) causes reduction in pathogen content by heating.
- Nitrification and denitrification, causing conversion of ammonia to nitrite and nitrate, and finally removal (loss) of N as nitric oxide (NO), nitrous oxide (N_2O) and/or molecular nitrogen (N_2).

Adequate aeration of liquid effluents (slurries) involves dissolving enough oxygen into the substrate to replace the anaerobic (chemically reducing) environment by aerobic conditions for microbial activity [8]. As a result, organic matter, characterized by BOD, will be rapidly oxidized to CO_2 and water.

The aerator *efficiency* is often used to indicate the energy used per unit of oxygen supplied; an efficient aerator delivers oxygen more cheaply than an inefficient one. Aerator efficiency is an important consideration as aerators account for most of the energy use of the process. Efficiency is often expressed as kg O_2 per kWh electricity consumed. Aerators are considered efficient if they achieve a performance of more than 1 kg O_2 per kWh; those of more than 5 kg O_2 per kWh, such as bubbler systems, are very efficient. However, bubbler systems are not commonly used for livestock slurries because of their low *specific aeration capacity* or *aeration intensity*. This is the amount of oxygen delivered per unit of reactor volume per unit of time (kg O_2 m^{-3} $hour^{-1}$).

Aeration for N removal. When slurry is sufficiently aerated, aerobic microbial activity dominates and free oxygen, rather than chemically bound oxygen, becomes the final electron acceptor. The relatively strong oxidizing environment leads to a more complete breakdown of organic compounds, with water, CO_2 and other simple molecules as final products. In this way, many of the organic compounds related to offensive odours are removed. The value of aeration in reducing offensive odours is widely accepted and has been demonstrated by many workers using olfactometric method [10, 11]. As the process advances and the more digestible material is consumed, BOD and COD contents decrease (at a diminishing rate) and the slurry becomes more stable. The presence of aerobic conditions also leads to a reduction in pathogen numbers as many of these are strictly anaerobic [8].

[7] According to Taiganides [9], BOD is the most widely used parameter in estimating the water pollution potential of organic waste. BOD is the measurement of the amount of oxygen utilized by micro-organisms to biochemically oxidize the organic component of the waste over a period of time at a specified temperature. A measured volume of the waste sample is placed in a BOD bottle that is filled with distilled water saturated with dissolved oxygen (DO) and to which specific chemical nutrient solutions have been added. The DO level is measured initially and after 5 days of incubation at 20°C. The difference in DO levels divided by the volume of the sample is the BOD_5 of the waste. The BOD_5 of a slightly polluted stream is less than 10 mg $litre^{-1}$.

The fate of slurry N during aeration is of particular importance. The general sequence of transformations observed can be summarized in four steps (Figure 6.1). Considering the important role of NH_4^+ in these transformations, it should be taken into account that most of the NH_4^+ in animal slurries is formed by hydrolysis of urinary urea (mammals) or breakdown of uric acid (poultry), the main N components in excreta. Hence, mineralization of organic N (step A) is a minor source of NH_4^+ in slurry. Hydrolysis of urea is catalyzed by the enzyme urease and generally completed within one day after urine production. Breakdown of uric acid is catalyzed by the enzyme uricase and takes more time (Chapter 5). Denitrification is an anaerobic process. This means that N removal from slurry is most effective when aerobic and anaerobic conditions alternate (intermittent aeration).

$$\text{Organic nitrogen} \xrightarrow[\text{bacteria}]{\text{Mineralization}} NH_4^+ \qquad \text{(A)}$$

$$NH_4^+ + \text{oxygen} \xrightarrow[\text{bacteria}]{\text{Nitrosomonas}} N0_2^- + 2H^+ + H_20 \qquad \text{(B)}$$

$$N0_2^- + \text{oxygen} \xrightarrow[\text{bacteria}]{\text{Nitrobacter}} N0_3^- \qquad \text{(C)}$$

$$N0_3^- + H^+ \xrightarrow[\text{bacteria}]{\text{Denitrifying}} N_2 \text{ or } N_20 \qquad \text{(D)}$$

FIG. 6.1. Sequence of N transformations in aerated slurry.

In the absence of nitrification (steps B and C), large NH_3 losses are likely, particularly if air flow rates are high. More than 3 days of treatment, with an aeration level of more than 1% of the saturation value for dissolved oxygen, enables development of a population of nitrifying bacteria [12]. Denitrification (step D) will occur after ceasing aeration. This is sometimes referred to as the anoxic phase. Denitrification may also occur simultaneously with nitrification if the aeration level is kept close to the minimum for nitrification.

Nitrogen in animal slurries causes at least three different types of pollution, *viz*. emissions of NH_3 and N_2O to the atmosphere and leaching of nitrate to groundwater and surface water. Emission and subsequent deposition of NH_3 is closely associated with the acid rain problem, the eutrophication of natural ecosystems, and the formation of airborne particulates that can be a human health hazard (Chapter 5). Ammonia emission from fields spread with animal waste has received particular attention in continental Europe. Aeration of slurry causes a temporary increase of slurry pH [13, 14] and this will stimulate NH_3 volatilization. In addition, aeration may increase NH_3 emission by stripping out NH_3 from the slurry, particularly if excessive air flow rates are used.

Nitrous oxide is a greenhouse gas with a high Global Warming Potential, *viz*. 296 times that of CO_2 for a time horizon of 100 years [15]. It is produced by nitrification and denitrification and, therefore, its production may be strongly increased by aeration of slurry, as reported by [16, 17].

Aerobic treatment of slurry to remove NH_4^+ as NO, N_2O and N_2 (following a nitrification-denitrification route) has been proposed to alleviate the problem of nitrate leaching in Brittany, France [16, 18]. However, it is questionable whether this technology should be recommended for this purpose. Nitrate leaching is a complex process that depends on the rate and timing of N application rather than on the source of N [19]. Hence, slurry aeration should be accompanied by a number of other measures to reduce nitrate leaching. In addition, slurry aeration stimulates harmful gaseous N losses, and the lower content of N in the final products probably will increase the environmental problems related to over-dosing of P and heavy metals [18]. All these aspects should be taken into account before propagating a rather expensive slurry processing technology like aerobic treatment.

6.3.1.2. Performance and experiences

Laboratory experiments provided data for the development of mathematical equations describing changes in the characteristics of aerated pig slurry. Thus:

$$BOD_{5(effluent)} = 1.568/R + 0.152*BOD_{5(influent)}$$

where R is the mean treatment time (days), and $BOD_{influent}$ is the biological oxygen demand of the fresh slurry (g $litre^{-1}$).

Thus, after 5 days of mesophilic[8] aeration, pig slurry with 10% DM and a typical BOD of 35 g $litre^{-1}$ would have a BOD of 5.6 g $litre^{-1}$, or *ca*. 16% of the original value. Odour panel assessments have shown that 2–3 days of mesophilic aeration result in offensive pig slurries becoming inoffensive [20].

Béline and coworkers [18] monitored three slurry processing plants on commercial pig farms in Brittany, France (Figure 6.2 and Figure 6.3). Slurry treatment on these farms was as follows: Farm 1: intermittent aeration, followed by sedimentation of aerated slurry. Farm 2: mechanical separation of raw slurry, followed by intermittent aeration of the liquid fraction and sedimentation of the aerated slurry. Farm 3: mechanical separation of raw slurry, followed by intermittent aeration of the liquid fraction and mechanical separation of the aerated slurry. The raw slurries had a low DM content (*ca*. 40 kg total solids per ton). This indicates considerable additions of water (cleaning water, spilt drinking water, rain). Figure 6.2 shows the N and P balances of these treatment systems.

[8] operating within a temperature range of 20–45°C

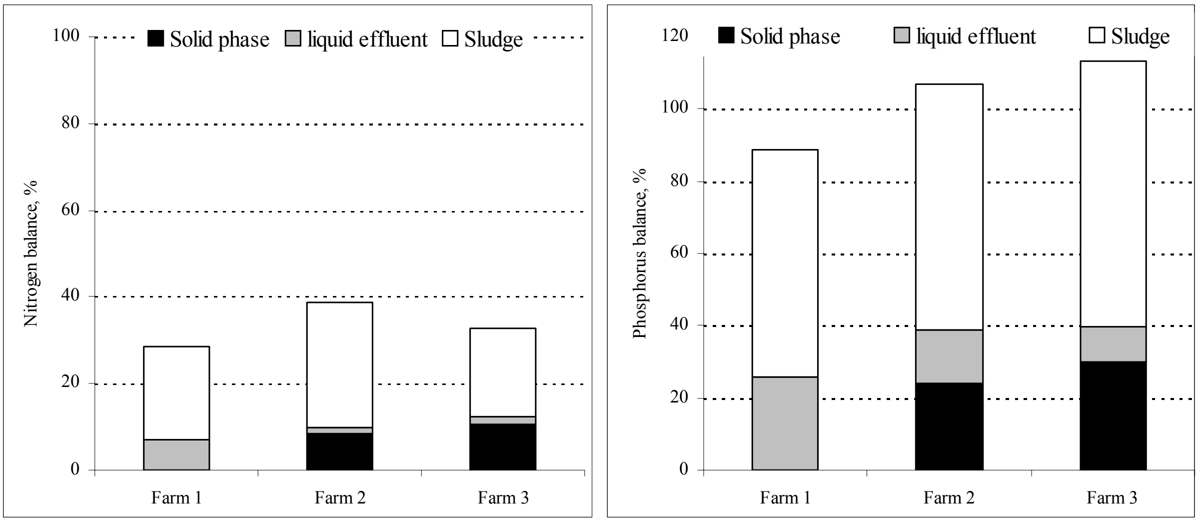

FIG. 6.2. Nitrogen and P balances of three farm-size pig slurry plants with aerobic treatment in Brittany, France [18].

All these systems of slurry treatment caused a removal (loss) of 62–72% of the initial amount of total N, apparently *via* gaseous losses of NH_3, NO, N_2O and N_2 (Figure 6.2). These losses were stimulated by aeration and although the overall objective of these treatment systems was to reduce nitrate leaching, harmful gaseous N losses should also be considered in their evaluation. However, no information was provided about these pathways of N loss.

Mechanical separation before aeration (farms 2 and 3) removed 8–10% of initial N and 25–30% of initial P (Figure 6.2). The volume of the solid fraction was 4–5% of the initial slurry volume. Sedimentation (farms 1 and 2) and mechanical separation (farm 3) of aerated slurry concentrated 20–29% of initial N and 63–73% of initial P in the sludge (*i.e.* in 35–40% of treated slurry). The liquid effluent (clarified supernatant) contained 2–7% of initial N, and 10–25% of initial P. Hence, the treatment caused large losses of gaseous N and concentration of P (and heavy metals Cu and Zn) in the sludge and to a lesser extent in the solid fraction. The latter increases the risk of overdosing P, Cu and Zn on agricultural land and related environmental problems.

FIG. 6.3. View of a pig slurry treatment plant as installed on a typical farm in Brittany (France).

6.3.1.3. Recommendations on this technique

Aerated lagoons. A simple low-cost system for aerobic slurry treatment, where odour abatement is required, is the aerated lagoon. Lagoons exist on many livestock farms and the installation of an aerator represents a relatively small investment.

Continuous aerobic treatment systems. There are now examples of continuous aerobic treatment plants (pilot and full scale) for livestock waste in Europe, especially in France and The Netherlands [17, 18]. Costs of operation are rather high (*ca*. € 7–17 t^{-1} in the study in The Netherlands) and should be covered by the economical benefits of manure treatment [17]. In recent years, small pilot plants have given way to full-scale units handling more than 100 tons per day.

Use of aerobic trickling biofilter. The use of technology where effluent passes over packing covered with active biomass, is common in the final stages of wastewater treatment in municipal sewage works. This method of aeration is less attractive for farm effluents owing to (a) the higher organic loading leading to the risk of blockage and (b) the problem of achieving the required amount of oxygen transfer. Nevertheless, because of the simplicity of the system, it remains attractive and several pilot schemes have been developed to further explore the best way of operation. It can be operated in one of two ways: naturally ventilated and forced ventilated. The former relies on air percolation.

Benefits and disadvantages of aerobic treatment. The claimed benefits of aerobic treatment of liquid manure (slurry) mostly relate to the reduction in a wide range of environmental impacts:

- reduction in emission of offensive odours, both during subsequent manure storage on the farm and after land-spreading;

- considerable reduction in methane (CH_4) emission [17];
- reduction in the concentration of inorganic N (NH_4^+) in the manure and, possibly, in nitrate leaching after field application of the manure;
- significant reduction in the concentration of reactive organic matter (BOD_5 content);
- sanitization of the slurry by inactivation of certain pathogens that are strictly anaerobic, such as Enterobacteriae (*Salmonella*, *Escherichia coli*), faecal s*treptococci* and some viruses;
- aerobically treated manure is also claimed to be less harmful to foliage (better nutrient utilization and performance of plants).

However, some *disadvantages* also have to be considered:

- increase in gaseous N losses, in particular NH_3 and the strong greenhouse gas N_2O;
- a lower N/P ratio in the end product and, hence, a lower value as a fertilizer for crops (Chapter 7) and a higher risk of accumulation of P and heavy metals in the soil.

6.3.2. Anaerobic digestion

6.3.2.1. General principle

Anaerobic digestion is one of the most important treatment measures available for animal manure and other organic wastes, which allows the production of a universal energy carrier, CH_4. The use of anaerobic fermentation for waste treatment is of large traditional importance, with several million small scale biogas plants in the Peoples Republic of China, India and other Asian countries. But in Europe, the development of this technology stagnated, possibly because it had to compete with the good infrastructure for the distribution of fossil fuels and electricity. Therefore, it is probably easier to promote anaerobic digestion in remote and rural areas in developing countries, which have less developed infrastructure for energy.

After the oil crisis in the 1970's and the high oil prices in recent years, the interest in renewable energy increased again for a while in most parts of Europe and so did the interest in biogas plants.

Anaerobic digestion is most easily and commonly carried out with pumpable slurries, although more recently, high solids content (20–40% DM) plug-flow reactors have been developed. Although the optimum dry matter content of slurries for anaerobic digestion is 6-8%, it is likely that the majority of cattle and pig slurries could be digested successfully, provided that excess bedding material was excluded. One of the products of the process is biogas, a mixture of 60–70% CH_4 and 30–40% CO_2.

6.3.2.2. Performance (biogas production) and experiences

Anaerobic degradation of organic substances to the most reduced form, CH_4, is a purely microbial process. The energy released during the degradation steps, which was originally stored in the substrate, is predominantly recovered by the CH_4 formed:

33 g organic material (C:H:O) = 22 g CO_2 + 8 g CH_4 + 3 g biomass

For an estimate of gas yield, the following equation can be used:

$V(CH_4) = 0.35\ (COD_{inf} - COD_{eff})*Q$

where COD_{inf} and COD_{eff} are the chemical oxygen demand values (kg m^{-3}) of the influent and effluent, respectively, Q the influent flow rate (m^3 day^{-1}), and $V(CH_4)$ the CH_4 production rate (m^3 day^{-1}) at standard pressure and temperature.

In practice, the CH_4 production potential of livestock slurries is assessed on the basis of the content of volatile solids (VS) in the slurry and empirical standards for the production of CH_4 per kg of VS. Typical gas yields for some organic substrates are shown in Table 6.2. Co-fermentation of cattle slurry with different quantities of fodder sugar beet resulted in very high biogas and CH_4 yields, due to the high content of easily fermentable organic matter. This substrate and other non-fibrous and not lignified plant materials are ideal co-ferments for animal farms with no hygienic risks compared to other organic wastes [21]. However, the shortage of feeds and forages in most Asian countries generally will not allow the use of the listed products for co-fermentation. In addition, an important side-effect of co-fermentation that requires attention is that it increases the amount of digested waste (and waste nutrients) that finally is available for spreading on the land. This may increase pollution problems, particularly in regions with manure surpluses or poor manure management.

TABLE 6.2. TYPICAL BIOGAS YIELDS FROM VARIOUS TYPES OF MANURE AND BIOMASS [21, 22]

Substrate	Range of biogas yield (litres/kg VS)	Mean biogas yield (litres/kg VS)
Pig manure	340–550	450
Cattle manure	150–350	250
Poultry manure	310–620	460
Horse manure	200–350	250
Sheep manure	100–310	200
Cereal straw	180–320	250
Maize stover	350–480	410
Fodder sugar beets	344–982	810[1]
		690[2]
Grass	280–550	410
Vegetable residues	300–400	350
Sewage sludge	310–640	450

[1] Thermophilic conditions, *i.e.* > 45°C; [2] Mesophilic conditions, *i.e.* 20–45°C; VS, volatile solids

Because COD is a measure of the mean oxidation state of organic carbon (MOC), it is evident that CH_4 yield also depends on MOC. The relationship between COD and MOC is given by the equation [23]:

$$MOC = 4 - 1.5 * COD/TOC$$

where COD is chemical oxygen demand and TOC total organic carbon. The mean oxidation state of the carbon (MOC) in different organic substances varies from –4 (in CH_4) to +4 (in CO_2). The closer MOC is to –4, the higher the CH_4 yield. The COD, TOC and MOC values of the feed material can be used for an estimation of the theoretical concentration of CH_4 in the biogas:

$$\% \; CH_4 = 19 * COD/TOC = (4 - MOC)/8 \times 100\%$$

Georgakakis [24] took liquid manure from a large modern pig unit for lab-scale digestion trials. The loading rate was increased in a series of five trials from 1.8 to 5.3 kg VS per m^3 reactor per day by gravity settling of the manure. The CH_4 yield increased with loading but only to approximately 4.8 kg VS per m^3 per day, which appeared to be the 'optimum' under the prevailing conditions. The corresponding CH_4 yield was 1.09 m^3 per m^3 reactor volume per day.

Another example has been reported from Italy, where under good conditions, a CH_4 production of about 15–20 m^3 $year^{-1}$ (about 25–35 m^3 biogas $year^{-1}$) could be achieved per 100 kg of pig live weight [25].

Biogas production typically varies from 0.8–1.6 (mean 1.2) m^3 per livestock unit (LU) per day, with a CH_4 concentration of 60%. Performance depends on the amount of straw in the manure and on fermentation time. Biogas is stored in a gas-holder and is used in co-generation units or for heating. From 1 ton of manure with 20% total suspended solids (TSS) and 50% straw, 20–25 m^3 of biogas can be produced with a total energy value of 100–125 KWh. By utilization of this biogas in co-generation units, 35–40 KWh of electricity and 55-75 KWh of heat energy can be generated [3].

6.3.2.3. Recommendations on this technique

Anaerobic lagooning. Anaerobic lagooning, as distinct from anaerobic digestion, does not have biogas production as a main objective. Some systems do include covers allowing collection of gas but with the inevitable (rather low) ambient temperatures, gas yields tend to be modest. Anaerobic processes offer a good conversion of organic carbon to CH_4, thus reducing organic matter and retaining a large fraction of the N and all the P in the end product, which is suitable for land application. Operating slurry lagoons as a treatment facility is more common in countries with a warm climate.

Co-processing and centralized facilities. The limitations of a small farm-based digester can be overcome in larger operations that include co-processing with other organic materials enabling:

- more efficient digestion of some biomass materials;
- easier handling of blended wastes;
- improved nutrient balance and utilization;
- additional income by charging gate-fees to take external wastes.

These benefits can be greatly increased with the large-scale production approach of centralized plants serving several farms along with the local community and food industry as well. This approach has been followed in several parts of Europe, in particular in Denmark where annual biogas production from such installations exceeded 2 million m^3 in 1994.

A plant built by a Danish firm at Cannington in Somerset is designed for a throughput of 200 tonnes of livestock slurries and other organic wastes per day and operates at mesophilic temperatures, plus pasteurization. Capital cost is reported to be *ca.* £ 4 million. Economics of such plants depend on payment of 'gate fees' on non-agricultural wastes, which may constitute up to 25% of plant throughput. Large Centralized Anaerobic Digesters (CAD) are seen by waste disposal contractors as an avenue for disposal of liquid organic wastes, because they are discouraged to dispose these wastes to landfills under the EU Landfill Directive. However, this approach presents considerable logistical problems of slurry transport to the central plant and transport of digested slurry back to farms for spreading on the land.

6.4. SOIL-BASED PROCESSES

6.4.1. Soil filters

6.4.1.1. General principle, theory and literature review

The underlying assumption in land application of animal manure is that the soil-plant ecosystem has the capacity to either immobilize or transform the manure components, thus avoiding pollution of groundwater and surface waters. Land application is the oldest system of waste disposal. According to [26], land is a gigantic bio-conversion system, developed during millions of years, and able to bio-degrade animal and plant wastes to become part of the soil. Land application serves two objectives: (i) waste disposal; and (ii) recycling of waste components. Fuller and Warrick [26] proposed the terms *land treatment* and *land utilization*. Land treatment involves the use of soil as a means of treating waste, while land utilization serves two objectives, *viz*. waste disposal and utilization of a valuable resource. Land treatment is based on the physical, chemical and microbiological interactions between the components and micro-organisms of soil and waste. Soil-based systems are in use for the disposal of many different types of organic waste, such as animal manures, municipal sludge, oil mill wastewater and meat processing effluents.

6.4.1.2. Performance and experiences

The movement of manure through soil results in a high degree of purification as long as the capacity of the soil is not exceeded. Purification is the result of physical separation and biological activity which actually mineralizes and utilizes the nutrients contained in the manure. These features are being used in the construction of soil filter systems.

A media filter has been constructed by [27] to treat swine wastewater after storage in an anaerobic lagoon. The media consisted of a tank filled with marl gravel. This media filter removed 54% of COD and 50% of TSS. Removal efficiency for total P ranged from 37 to 52% and up to 24% of total N was converted to NO_2^- and NO_3^-, which were further denitrified in constructed wetlands.

Boiran and coworkers [28] described N removal from pig slurry based on a forced nitrification step using gravel columns. Depending on the content of the column (calcareous or siliceous gravel), they obtained 4 to 38% removal of total N and 64 to 98% oxidation of NH_4^+-N (into NO_2^-- and NO_3^--N).

In Hungary, a four-stage soil filtering system has been studied by [29] for the treatment of very diluted pig slurries (0.4–0.6% DM). This simple low-cost system was run with a prefilter of straw. The filter beds consisted of wood shavings, gravel and sandy soil. The load of the system was 2.5–5.0 m^3 day^{-1}. Its overall removal efficiency for COD was 43-76%, for BOD 46–88%, and for total suspended matter 58–99%.

A soil treatment process, called barriered landscape wastewater renovation system (BLWRS), has been developed in the USA and consists of a mound of soil underlain with an impermeable barrier and drainage system [30].This creates an aerobic zone in the top of the BLWRS and an anaerobic zone next to the impermeable barrier. During a two-year study of this soil filter system with liquid dairy waste, it was capable of reducing COD and N by 90% or more and P by 99%.

The SOLEPUR system

The 'Solepur' system has been set up in 1990 in Brittany, France, to determine the purification performance of a natural soil filter system for the treatment of raw pig slurry [31].

It is based on a unique design, *viz*. a field-scale macro-lysimeter of 3280 m^2. The system (Figure 6.4.) involves the following operations:

- applying high rates of slurry (1000 m^3 ha^{-1} $year^{-1}$) to the managed field;
- collecting and treating the nitrate-rich leachate produced;
- applying the treated water to other fields.

In a 5-year study, N removal efficiency, and the behaviour of P, potassium (K), and heavy metals have been assessed. Large volumes of slurry were applied to the managed field, with an average annual load of 4900 kg N ha^{-1}, 1593 kg P ha^{-1} and 3304 kg K ha^{-1}. The process removed 99.9% COD, 99.9% P and *ca*. 90% N from the slurry. The leachate contained a very low concentration of organic matter, but high NO_3^- levels, resulting from the oxidation of slurry N in the soil. A number of ions, including NO_3^--N, K^+, Ca^{2+}, Cl^- and SO_4^{2-} were heavily leached at rates ranging from 400–600 kg ha^{-1} $year^{-1}$ [32].

6.4.1.3. Recommendations on this technique

This approach is possibly of more immediate interest as an option for treatment of dirty water. In these latter studies, percolation systems constructed on a permeable soil, and an overland flow system constructed using an impermeable soil, have shown considerable promise. Percolation systems working on a continuous basis reduced BOD and NH_4^+-N by >90%, at dirty water application rates of 2 or 8 mm per day. Overland flow systems, working on a batch-flow basis, significantly reduced BOD (>85% removal in 10 days) and NH_4^+-N (>90% removal in 10 days).

Most of the land treatment systems have been studied for a relatively short period. A point of concern is their long-term effectivity, because the capacity of the soils to accumulate and retain P and other minerals is limited [33]. In addition, some manure components, like heavy metals, antibiotics, and veterinary medical residues may damage the purification capacity of the soil.

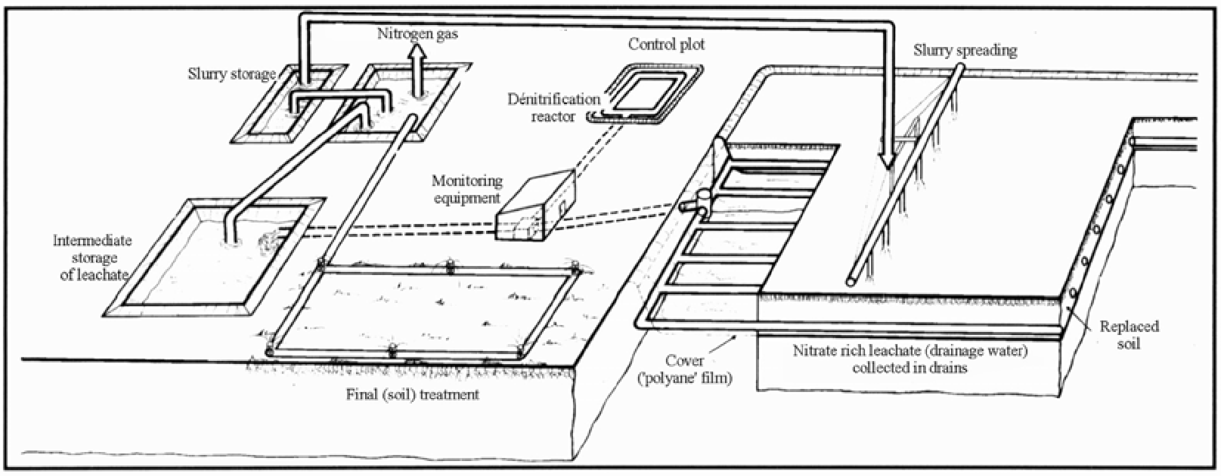

FIG. 6.4. The Solepur field treatment plant.

6.5. PRODUCTION OF ORGANIC FERTILIZERS BY COMPOSTING AND PELLETIZING

Composting of dry manures and organic wastes has become increasingly popular. This possibly results from the claimed environmental benefits, the commercial interest in composted material and the reduction in landfill charges as a result of diverting the waste to an alternative route. Composting of solids may sometimes involve the use of slurry as a source of N. Also, composted manures will pose a much reduced risk of pollutant run-off during storage or following land application, as a result of bio-conversion of organic compounds that otherwise may contribute to such pollution.

Composting may result in the following benefits:

- reduction in manure volume and mass, due to decomposition of organic matter and loss of CO_2 and water;
- stabilization of manure, resulting in reduced emissions during storage and following land application;
- inactivation of weed seeds and some pathogens;
- changed nutrient availability by mineralization and gaseous losses (*e.g.* NH_3); however, this can be both an advantage or disadvantage (Chapter 7);
- opportunity to develop alternative applications, *e.g.* recycling of composted material as livestock bedding, use as horticultural growing medium and soil amendment, use in gardens and parks;
- scope to transport surplus nutrients to regions deficient in organic matter and plant nutrients;
- associated with a positive public image for waste recycling and environment protection.

There are three main composting systems:

(1) windrow,
(2) static pile with forced ventilation,
(3) in-vessel.

Control over the composting process increases from windrow to static pile and in-vessel composting, as does the capital costs. Labour costs decrease in the same order and the overall operation costs mainly depend on the costs of labour and energy.

6.5.1. Windrow composting

The raw material is piled in long rows (windrows) and turned at intervals using mobile equipment like tractors with front loaders or compost-turners, machines specially designed for compost turning. The most common method, the conventional windrow, is aerated through natural ventilation (convection and diffusion), and also during turning, which is also required for more homogenous composting. This process requires an extensive area. The base of this area can be compacted soil, but ideally it is concrete with the facility to collect any leachate. In regions with high rainfall, leachate production can be reduced and improved control of composting achieved by roofing the composting area.

6.5.2. Static pile composting

This system of composting uses an active aeration system. Perforated pipes are laid on the floor or in floor channels and covered with porous material like straw, wood chips, *etc*. This stimulates efficient distribution of air. The raw material is then piled on the base and covered with a layer of matured compost to provide thermal insulation and partial odour removal. Aeration, controlled by temperature feedback, is used to sustain the pile in an aerobic state, to maintain the temperature of the pile and to control the moisture content. Heat is produced by aerobic decomposition of organic matter and causes evaporation of water (and compost drying).

6.5.3. In-vessel composting

This system is used to ensure homogeneous composting, inactivation of pathogens and odour reduction. In-vessel composting includes temperature control and is usually a multi-stage process. Pre-composting or full composting is achieved in the first stage in a bioreactor, and the final composting and maturing in windrows. The most common types of reactors are horizontal and vertical plug-flow and, also, an agitated bin reactor. Some systems incorporate computer control of temperature and oxygen levels. The quality of exhaust gases is often improved by passing them through a biological filter for odour and NH_3 removal. This type of composting, being well controlled and thoroughly mixed, is faster than the previous systems, but the more complicated control and processing mechanisms are expensive and require costly maintenance.

While moisture content decreases from about 70% to less than 30% and organic matter content from about 75% to 50%, the concentrations of P and metals in the DM increase. By oxidizing the bio-degradable carbonaceous compounds to CO_2, the compost is biologically stabilized, *i.e.* when stored without aeration and rewetting, it does not generate any odorous compounds and its biological activity is minimal. This also means that the potential BOD emission, *e.g.* in leachate from stored material is greatly reduced. Odour is produced mostly at the beginning of composting, when odoriferous compounds already contained in the raw material are released in the exhaust gases by the increased temperature and forced aeration or turning. To minimize odour emissions, the windrows are covered with mature composted material or the air sucked from static piles is filtered through a biological filter.

The cost of in-vessel composting would be prohibitive for farmers if, for example, a system which provides continuous composting with internal mixing and biofiltration of exhaust gases, was to be used. The indicative costs of the composting plant would be around £ 0.75 million and depending on the waste stream, treatment costs of one ton of manure could be in excess of £ 50. However, there is potential for poultry manure to be used as an amendment in such systems to assist with the composting of the main waste stream. Since the treatment of slurry would require the addition of bulking material, like straw, wood chips, *etc*., the advantage of reduced waste weight and volume due to composting would be compromised. For livestock slurries, the necessary addition of DM to reach the necessary solid concentration (25–35%) can be so high that composting may become impractical. For example, starting with one ton of livestock slurry of 5% DM content, the raw material requires an addition of 0.3 ton of dry bulking material in order to obtain a mixture with 25% DM.

6.5.4. Solids pelletizing

Some more novel options, arising from or linked to manure processing, also need to be considered. In the State of Delaware, USA, the world's largest chicken manure pelletization plant has processed about 60,000 tons of chicken manure since it opened in July 2001. The plant was designed as a solution for local poultry farmers who needed to remove waste from their facilities. Most of them had no option but to spread it on fields according to their Nutrient Management Plan, or store it in special leak-proof structures. A large farming company researched different methods of addressing the surplus manure, including incineration (an idea that was abandoned due to the costs and complexity of meeting emission restrictions) or a composting facility (which proved to have too many logistical problems). The pelletization plant, 'Perdue Agr-Recycle', which handles manure from both Delaware and Maryland, was chosen because the waste could be transported easily before and after processing and it produced a marketable product. Most pellets are sold directly to farms or other outlets (*e.g.* golf courses) in 1 ton containers, but smaller amounts are sold *via* the retail trade, giving rise to products such as 'Fertile GRO' and 'Cockadoodle DOO' in the USA. Such products have been available for many years in Europe and an example is 'Rooster Booster' currently selling at £ 3.48 per 7 kg bucket through B&Q. Concern about odour emissions from the USA plant seem to have been allayed by almost two years of operation with few complaints.

6.6. CONCLUSIONS AND SOME RECOMMENDATIONS FOR ASIAN COUNTRIES

6.6.1. Physical treatment

A wide range of possibilities include sedimentation, separation, filtration, screening, centrifugation, and solar drying. Some of these techniques involve high investments and operation costs (centrifugation), while others could be implemented using local resources (sedimentation, screening, solar drying). Typically, the main advantage is to obtain easier handling of liquid manure and solids for direct application or composting. Filtration processes would be limited to diluted wastewaters (less than 1% DM).

6.6.2. Biological treatment

Here again, a wide range of possibilities and technologies exists including aerobic reactors, anaerobic lagoons with biogas collection, farm-based biogas units, and centralized biogas plants. Most of these systems are effective for odour abatement, pathogen control and nutrient removal (N). Disadvantages are the high costs of operation (for aerobic systems), the land area needed for anaerobic lagoons and sludge disposal problems. There is also a need of effective use of the biogas produced by some of these systems.

6.6.3. Production of organic fertilizers

This is mainly related to large compost plants and farm-based compost units. In the first case with large-scale operation, there is a need for machines for mixing, aerating and handling, while the individual composting unit requires more local labour. Up to now, the main drawback of this approach has been the uncertainty of markets for the compost produced.

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7. FIELD APPLICATION AND UTILIZATION OF MANURES

Much of the nitrogen (N), phosphorus (P) and potassium (K) in livestock diets is excreted in faeces and urine (Chapter 4). Manure contains these plant nutrients as well as the other major nutrients and trace elements. Manure was the only important source of crop nutrients until the 1850s when superphosphate appeared as the first commercial chemical fertilizer. In recent times, chemical fertilizers have been widely available at low cost and the dependence of crop production on animal manures has decreased. In spite of this, manure production has increased and, in some areas, intensive animal production has become associated with excessive production of manures. In intensive crop and livestock production systems, livestock manure has tended to be considered as a 'waste', to be disposed of as cheaply and conveniently as possible. The rapidly increasing production of animal manures in Asian countries, where arable farmers have good access to low-cost chemical fertilizers, is thus causing serious pollution of the environment, in particular of water resources. European experience suggests that utilization of these manures on cropland and grassland is a cheap and sustainable solution for this problem [1]. However, a challenge will be to persuade governments, researchers and farmers that this is also true for Asian conditions?

Many traditional agro-ecosystems use animals to collect and concentrate plant nutrients. In this way, the management of manure causes a transfer of plant nutrients from grazing to cropping areas and can result in a substantial contribution to the crop nutrient supply. However, this often results in declining soil fertility and degeneration of grazing lands [2]. The transfer and concentration of plant nutrients is also a problem in intensive livestock production: animal numbers are often excessive in some regions and land resources are not sufficient to utilize all the manure produced. Van Boheemen [3] detailed a case study on this problem in The Netherlands in the 1980s. Environmental legislation and financial incentives for removal of livestock farms have been necessary to reduce the problems described by Van Boheemen.

Animal manures are valuable when used carefully as fertilizer for crop production and maintenance/improvement of soil quality. Despite alarming reports on increasing manure production and regional manure surpluses in Asian countries [4], present-day production of the most important arable crops (rice, wheat, maize) is largely based on chemical fertilizers. Chemical fertilizers are cheap, easy to transport and to apply and have reliable effects on crop growth and production. However, many long-term experiments with rice in monoculture and rice-wheat rotations show a decline in yield over time [5–8]. Possible causes of this are: (1) a decrease in soil organic matter content and related chemical and physical soil quality parameters, and (2) negative balances and deficiencies of P, K and other secondary and micronutrients [6]. Proper use of livestock manures on crop land may solve these problems and reverse the yield decline observed. Hence, this most obvious solution of the manure surplus problem, *viz*. recycling of manure nutrients for crop production, should receive much more attention in national and international research programs on crop nutrition and soil fertility management.

The potential fertilizer value of the manure produced by a cow over the winter housing period based on recent price for N, P, K fertilizer, is approximately 40–50 US$. Of course, in many Asian countries the cows do not have access to grazing and are housed year-round. Therefore, the manure should be used, as far as possible, as a significant source of nutrients instead of chemical fertilizers. However, the costs of collection, storage and application are often likely to be higher than the value of the nutrients. Thus, at country level, land application of animal manures is likely to provide the cheapest solution of the pollution caused by manure accumulation (where pollution of water, soil and atmosphere and related deterioration of the quality of life is recognized as a public cost). However, for individual farmers, such increased costs can rarely be sustained without legislation and, possibly, some support.

7.1. MANURE NUTRIENT CONTENT

A serious problem of 'manure nutrient content' is the heterogeneity of manures, and, related to that, the difficulty of reliable sampling and chemical analysis. This is a problem in any part of the world and questions the benefits of manure analysis before field application. Moreover, chemical analysis is usually time-consuming and expensive.

The 'typical' nutrient contents of manures are of some interest and can be used for general planning purposes. Average nutrient contents of the most important manure types in Western Europe are presented in section 4.2 of this document. However, for reliable fertilizer planning, it is important to use local information and, where possible, to estimate the nutrient content of manure on a farm basis. The composition of manures is variable, this being true even for the manure of one animal category. Generally, the degree of dilution with water is the main cause of variability: the nutrient content per kg of manure dry matter (DM) is much less variable than per kg of 'fresh' manure. This means that manure DM content provides a good indication of nutrient content. The same is true for the electrical conductivity (EC), related to the content of salts. Based on these principles, simple methods have been developed to estimate the nutrient content of manures. In this case, the Hokkaido system for predicting the nutrient content of manures is outlined [9].

A series of equations (Tables 7.1–7.3) have been developed from multiple regression analysis of data collected in Hokkaido, Japan, based on measurement of EC and DM content. The latter is estimated by assessment of specific gravity, using a hydrometer. Whilst the specific calibration equations apply only to Hokkaido, the principle of the method may be adapted in other regions. Conventionally, the nutrient content is given as the element for N and as the oxide for P (P_2O_5) and K (K_2O), as for fertilizers.

7.1.1. Dairy cattle manure

An outline of the method for determination of the N, P2O5 and K2O content of solid manure, slurry and liquid manure from dairy cattle and pigs, based on EC and DM content, is shown in Table 7.1. The multiple regression equations used for predicting nutrient content of cattle manure are presented in Table 7.2.

TABLE 7.1. ANALYTICAL PROCEDURES FOR PREDICTING THE NUTRIENT CONTENT OF CATTLE AND PIG MANURE [9]

Parameter	Type of manure	Procedure
EC[1)]	Solid manure	Dilute the fresh sample 5 times with tap water. Shake for 30 minutes and measure EC by EC meter.
	Slurry	Dilute the fresh sample 2 times with tap water. Shake vigorously for a few seconds and measure EC by EC meter.
	Liquid manure[2)]	Measure EC directly by EC meter
DM	Solid manure	Dry the sample at 105°C for at least 24 hours.
	Slurry	Dilute the fresh sample 2 times with tap water. Shake vigorously for a few seconds and measure EC by EC meter. Then, measure specific gravity (SG) of the diluted samples after at least 1 minute following insertion of the SG meter into the sample. If SG is more than 1.03, dilute the sample again. If SG is less than 1.03, DM can be calculated by the following equation: DM = 218.96(SG–1) x dilution rate

[1)] Before EC measurement of solid manure and slurry, EC of the tap water should be measured; [2)] liquid manure is the liquid fraction of animal excreta draining from a tie-stall housing system.

TABLE 7.2. REGRESSION EQUATIONS FOR PREDICTING THE NUTRIENT CONTENT OF CATTLE MANURE [9]

Type of manure	Parameter (% of fresh weight)	Regression equation
Solid manure	Total N	0.0459EC + 0.0124DM + 0.1249
	NH_4^+-N	0.0256EC – 0.0153
	P_2O_5	0.0238EC + 0.0092DM + 0.0918
	K_2O	0.1341EC + 0.0071DM – 0.0041
Slurry	Total N	0.0314EC + 0.0172DM – 0.0553
	NH_4^+-N	0.0201EC + 0.0037DM – 0.0412
	P_2O_5	0.0069EC + 0.0119DM + 0.0090
	K_2O	0.0338EC + 0.0063DM + 0.0236
Liquid manure	Total N	0.0148EC – 0.0366
	NH_4^+-N	0.0086EC – 0.003
	K_2O	0.0235EC – 0.0268

EC, mS cm^{-1} (25°C); EC in this table = EC value of the sample – that of tap water; DM, weight %

There was a statistically significant relationship between the nutrient content of manure observed by chemical analysis and that predicted from EC and DM. For example, the close relation between the observed and predicted nutrient content of cattle slurry (Figure 7.1) demonstrates the soundness of this method.

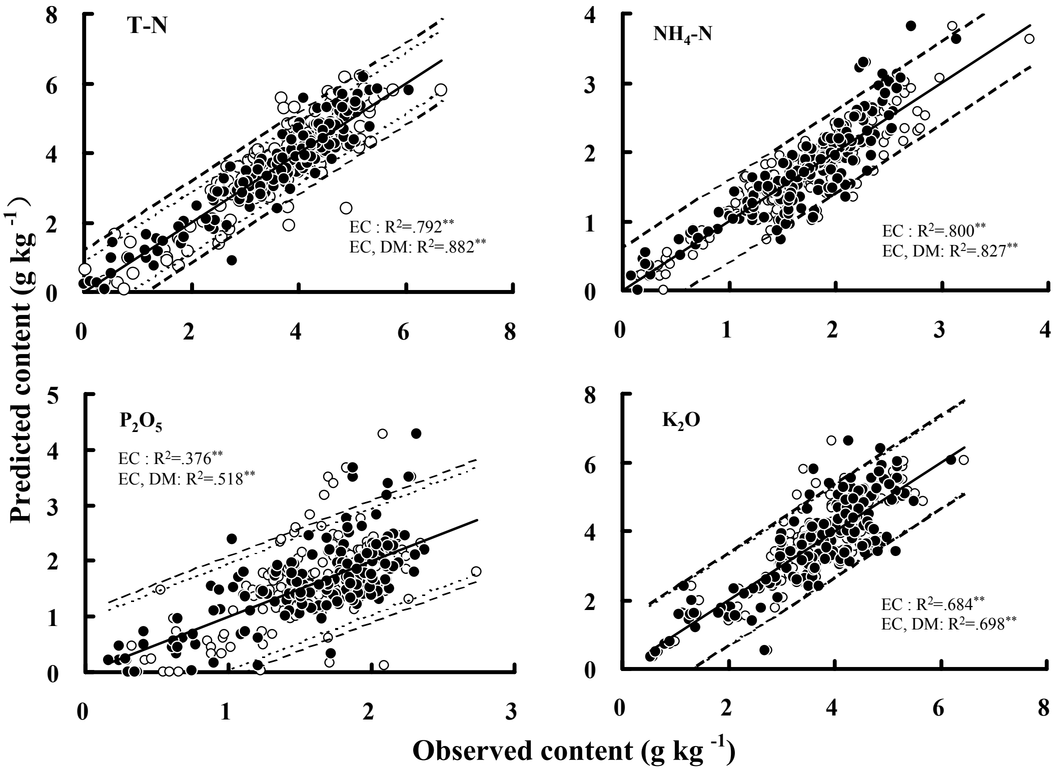

FIG. 7.1. Relationship between the nutrient content of cattle slurry observed by chemical analysis and the nutrient content predicted on the base of electrical conductivity (EC) and DM content [10].

○: Prediction by using only EC as a variable;

●: prediction by using both EC and DM in the manure as variables.

Long and short broken lines in the figure show confidence interval of 95% by multiple regression equation including only EC or both EC and DM as variables, respectively.

** Statistically significant ($P<0.01$)

7.1.2. Pig manure

The analytical procedures to estimate the nutrients content of pig manure are the same as presented in Table 7.1. Nutrient content of the pig manure is calculated *via* the regression equations shown in Table 7.3. A close relationship between the predicted and observed nutrient content in pig manure is shown in Figure 7.2, in particular, where both EC and DM are included in the regression equation. This is most obvious for total N and NH_4^+-N. This is convenient, because an accurate application of N is generally more important than of P and K. The rather poor prediction of P and K (from EC and DM) is less critical because a 'typical' value is often acceptable within an integrated nutrient management plan (with fertilizers and manures). In such cases, crop responses to fresh applications of P and K are unlikely, so the aim is to replace what is removed of these nutrients with the crop and maintain satisfactory soil P and K status only. The latter can be checked by occasional soil analysis.

TABLE 7.3. REGRESSION EQUATIONS FOR ESTIMATING NUTRIENT CONTENT OF PIG MANURE [9]

Type of manure	Parameters (% of fresh weight)	Regression equation
Solid manure	Total N	0.0771EC + 0.0285DM – 0.1538
	NH_4^+-N	0.0627EC – 0.033
	P_2O_5	–0.0453EC + 0.0748DM – 0.5757
	K_2O	0.0173EC + 0.0205DM – 0.0538
Liquid manure	Total N	0.0268EC + 0.0018
	NH_4^+-N	0.0252EC – 0.0111
	P_2O_5	0.0014EC + 0.0359DM + 0.0118
	K_2O	0.0210EC + 0.0250

EC, see Table 7.2; DM, weight %

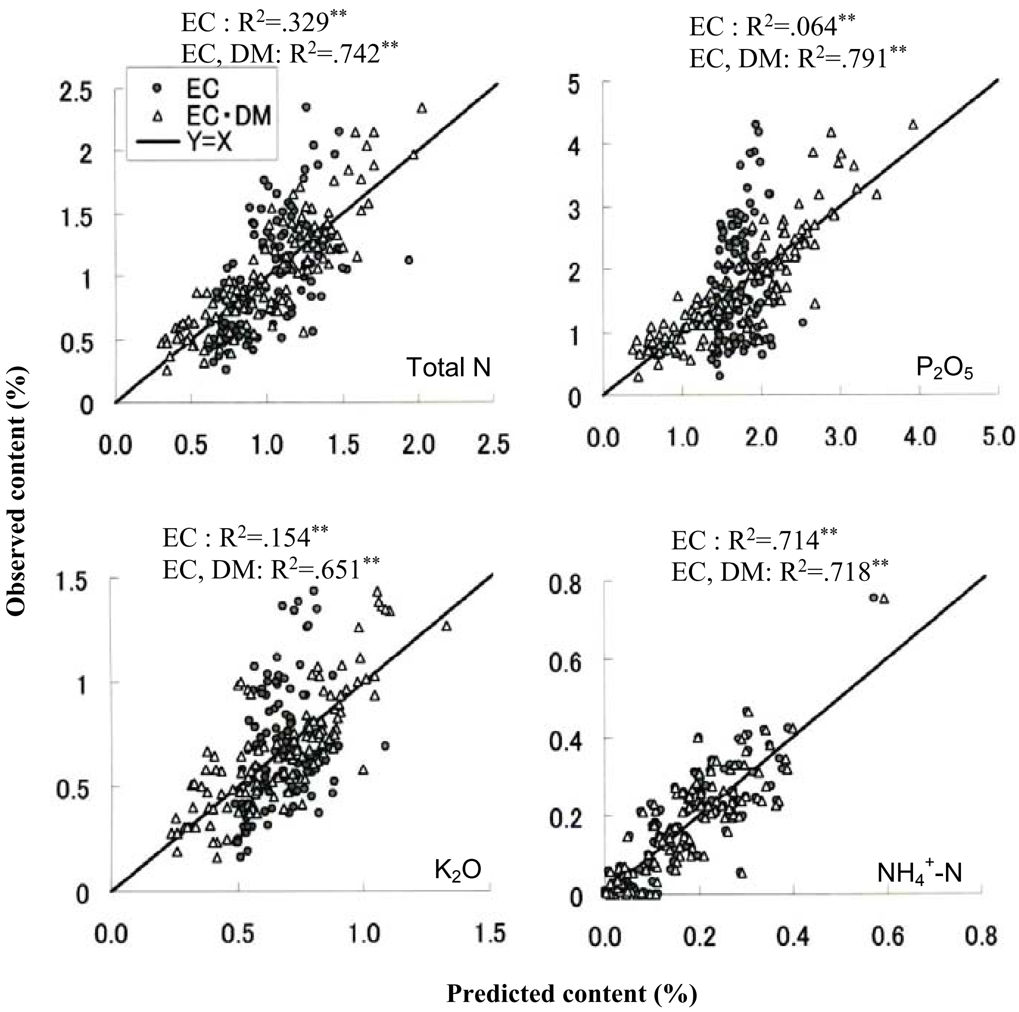

FIG. 7.2. Relationship between the nutrient content in pig manure predicted on the base of electrical conductivity (EC) and DM content and observed by chemical analysis [11]

● Prediction by using only EC as a variable; △prediction by using both EC and DM in the manure as variables.
** Statistically significant (P<0.01)

In the absence of easily accessible laboratory services for farmers in some countries in Asia, alternative sources of guidance need to be considered. The use of 'typical' nutrient content data of the type presented in Table 4.4 in this report has long been of some value to farmers in Europe, who have tended to prefer this approach to the cost and inconvenience of sending samples to a commercial laboratory for analysis. Of course, country-specific data on manure analysis would be preferable and it is recommended that a programme for analysing the most important manure types following a sound protocol for taking representative samples is undertaken, as soon as possible.

In the interim and, also in the mid-term future, it is recommended that sources of the on-farm analysis equipment are sought (details of suppliers in Europe and Japan can be provided by the contributors of this document). Some of these techniques have been tested by groups of farmers in the UK and Japan, with good results. Over half the farmers in the UK study indicated that they would be prepared to buy a slurry N meter or conductivity meter, although price at the time of the study appeared to be a concern with some farmers [12]. For all but intensive animal production units in Asia, the on-farm sampling equipment is likely to remain within the reach only of consultants.

7.2. ORGANIC MATTER IN MANURE AND SOIL AMELIORATION

Organic matter in manures plays an important role in promoting and maintaining good soil chemical, physical and biological fertility: the manure providing a source of stable organic matter and nutrients that encourage microbiological activity as well as plant growth. Manure DM, in slurries as well as in solid manures, generally contains *ca*. 70% organic matter [13].

Whereas the supply of nutrients to plants from manure is relatively predictable, its contribution to soil physical and biological fertility is not always consistent. This is dependent on the original content of soil organic matter. Thus, organic matter derived from applied manure will be very important (1) where the manure is incorporated into a soil of an organic matter content of less than *ca*. 5%, and (2) where soil physical properties are a major limiting factor for crop growth (Table 7.4). The decline of soil organic matter content has been mentioned as one of the possible causes of the yield decline observed in many long-term experiments with rice in monoculture or rice-wheat rotations using chemical fertilizers [6]. A good supply of organic matter to the soil is of particular importance in rain-fed production systems, where it contributes to soil structure, water holding capacity, cation exchange capacity and 'natural' supply of plant nutrients. Positive effects of regular manure applications on crop yield have been observed in a long-term experiment in a wheat-maize rotation in the North China Plain [14]. However, much research is still required to develop proper fertilization plans based on applications of both animal manures and chemical fertilizers. Special attention should be given to production systems with alternating flooded rice and dry-land crops and, hence, alternating anaerobic and aerobic decomposition of organic matter [7, 15]

When manure is applied to the surface of grassland soils, the manure solids will normally be incorporated by soil fauna, in particular by earthworms, and both this organic matter, as well as the related activity of soil fauna will have a positive effect on the physical properties of the soil.

TABLE 7.4. MAJOR FUNCTIONS OF MANURES APPLIED TO DIFFERENT SOIL TYPES [16]

Major function	Sub-function	Reclaimed land	Arable land		Paddy field	
		High content of OM in the soil**	Low content of OM in the soil*	High content of OM in the soil**	Low content of OM in the soil*	High content of OM in the soil**
Source of plant nutrients	N, P, K	o	o	o	o	o
	Micro nutrients	o	o	o	–	–
	Slow release fertilizer	o	o	o	o	o
	Plant hormone	o	–	–	–	–
Source of stable organic matter	Improvement of soil physical properties	o	o	–	o	—
	Cation exchange capacity	o	o	—	o	—
	Depressor of toxic substances	o	o	—	o	—
	Solvent of micro elements	o	o	—	o	—
	Buffering materials	o	o	—	o	—
Source of micro-organisms	Direct source of micro-organisms	o	—	—	—	—

OM, Organic matter; O, expectable for the function; -, Not expectable for the function; *, Less than approximately 5% organic matter; **, More than approximately 5% organic matter

7.3. IMPACT OF PERIOD, METHOD AND RATE OF MANURE APPLICATION ON NUTRIENT USE EFFICIENCY

Period, method and rate of manure application are very important for efficient utilization of manure nutrients and for minimizing environmental risk. This means: (1) applying the manure just before the start of crop growth and active nutrient uptake, at a rate that does not exceed the nutrient requirement of the crop, and by a method that limits NH_3 losses; (2) avoiding damage to the soil (*e.g.* compaction) and crop; and (3) considering requirements and costs of manure storage, manure application equipment, and manure processing (*e.g.* separation of solids and liquid). Again, the development of proper systems is an important research need for Asian cropping systems.

Generally, application time and spreading opportunity for manures are restricted according to land use (*e.g.* fallow, arable crops, paddy fields, grassland), type of crop and crop development, and weather and soil conditions (determining access and trafficability of the land). For example, on tillage land, the manure can be applied only before sowing or after harvest. Furthermore, nutrient supply from the manure, especially of N, will vary according to the form of the nutrients in the manure and their transformations. Manure total N consists of inorganic N (NH_4^+-N) and organic N (Table 4.4). Inorganic N is directly available for crop uptake unless it is lost by NH_3 volatilization or other N loss pathways. Organic N is only available after mineralization. In temperate climates, *ca*. 30% of manure organic N becomes available for crop uptake in the first year after application and the rest may gradually mineralize afterwards [17]. Most important for estimating the N value of organic manures is the fraction of NH_4^+-N lost by volatilization of NH_3. In extensive research in The Netherlands, NH_3 losses after surface spreading of cattle and pig slurry to arable land and grassland averaged *ca*. 70% of the amount of inorganic N applied [18, 19]. Typically, more than 50% of these losses were observed in the first three hours after application. The variation of these losses (range: 30–100% of inorganic N applied) and their low predictability favour the introduction of slurry application methods with much lower NH_3 losses, such as direct incorporation on arable land and injection techniques on grassland (Huijsmans *et al.*, 2001 and 2003). The availability of N in manures can also vary with application time. However, because of the wide range of soils, cropping and climatic conditions represented in Asian countries, there is an urgent need for research addressing these issues.

7.3.1. Manure application rate

It is generally accepted that manure application rate may significantly affect both nutrient uptake and crop yield following applications. Excessive application rates can give rise to negative effects on crop yield and quality [20], as well as increased risk of environmental pollution [21, 22].

7.3.2. Manure nutrient efficiency

In the context of considering future research needs in Asian countries, it is useful to explain how to determine and to express the efficiency of manure N use. This requires an experimental approach in which the effects of manure N and fertilizer N on DM and N yield in the harvested parts of the crop are compared. Such experiments include plots without applied N (control) and plots with only slurry and only chemical fertilizer N. The experiments also require ample supply of P and K to all the plots, to avoid crop response to manure P or K, rather than to N. A robust analysis of results requires determination of the N content of the harvested crop parts.

The efficiency of use of N from animal manure or chemical fertilizer can be expressed in two ways: (i) as the apparent recovery of N (ANR), which is the increase of the amount of N contained in the harvested parts of the crop, expressed as a percentage of that applied in manure or fertilizer, and (ii) as the apparent efficiency of N (ANE), which is the increase of crop yield (fresh or DM yield) per kg N applied in manure or fertilizer. The ANR and ANE values are therefore calculated from the respective differences in N uptake and crop yield between the manure or chemical fertilizer-treated plots and the untreated (control) plots [17].

The ratio $ANR_{manure}/ANR_{fertilizer}$, expressed in %, is called the ‘efficiency index of manure N for N uptake’. An efficiency index of manure N of 50% means that 100 kg of manure total N has the same effect on crop N yield as 50 kg chemical fertilizer N. This relates to the year of manure application. Similarly, the ratio $ANE_{manure}/ANE_{fertilizer}$, expressed in %, is called the ‘efficiency index of manure N for crop yield’. Efficiency indices are needed to include animal manures in fertilization programs.

Efficiency indices of manure P and K are much more difficult to determine, especially where background P and K status of the soil is already moderate or high. Generally, it is considered acceptable to take account of the total P and K content and assume close to 100% efficiency for manure P and K in situations with regular applications of animal manures [23].

7.4. INTEGRATED APPROACHES FOR MANURE AND INORGANIC FERTILIZERS

A fertilizing approach that integrates animal manure and fertilizer N is advisable because of the variability in manure analysis, and the difficulties of achieving high rates of manure N supply (because of the relatively low N content of most manures) and accurate manure spreading [24]. The aim should be to supply no more than 50–60% of the crops' expected N requirement from organic manure. A good alternative is to supply about 100% of the crops' expected P requirement. This strategy takes advantage of the additive effect of manure and fertilizer N, demonstrated in several studies [17, 25], and reduces the impact of variable N supply from the manure source. A typical N response curve for grass silage DM yield shows that a significant proportion of yield is obtained from soil N reserves (Figure 7.3). In this example, the grass continues to respond to extra N up to the optimum of about 100 kg N ha^{-1}. If half of this is supplied from manure, with the other half from inorganic fertilizer, *i.e.* 50 kg ha^{-1}, the farmer can be confident of the majority of the yield response to these sources. The slope of the response curve for yield at 100 kg N ha^{-1} is very small (shown by the horizontal arrow in Figure 7.3), so any variation in the manure N supply at this level is unlikely to have more than a minor effect on DM yield.

This type of integrated fertilizer-manure policy recognizes the difficulty of meeting the N requirements of crops by animal manure, not only on a short-term, single season basis, but also within the crop rotation. To rely on manures for a greater proportion of the crop N requirement within the rotation would inevitably result in significant P enrichment of the soil. The N:P ratio of the nutrient requirements of many cropping situations is in the region of 7–11:1, whereas the N:P ratio of manures is well below this at 2–6:1, using N_{total}, or 0.6–3:1, using readily available N (N_{RAN}, *i.e.* NH_4^+-N in cattle and pig manures, and NH_4^+-N + uric acid N in poultry manures) (Table 7.5). This is not an issue where there is a need to build up soil P reserves, however, where soil P has already attained a satisfactory level, care must be taken to avoid excessive enrichment, particularly with regular use of pig or poultry manures.

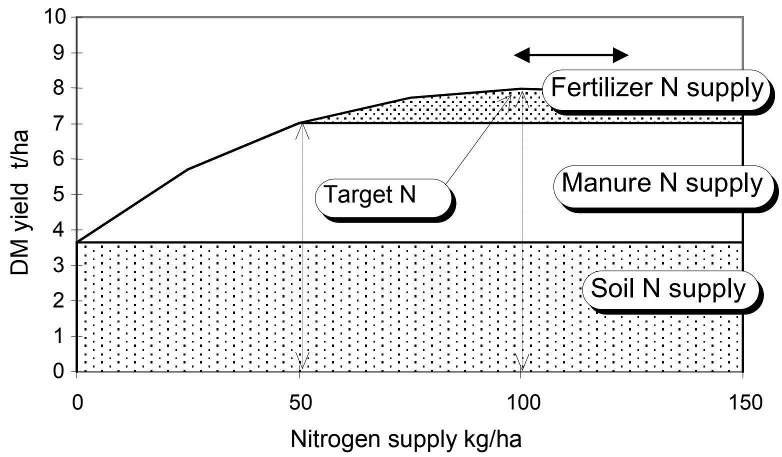

FIG. 7.3. Supplying grass silage N requirement from fertilizer and manure sources.

TABLE 7.5. RECONCILING THE N AND P SUPPLIED BY ANIMAL MANURES (DERIVED FROM MANURING RECOMMENDATIONS IN THE UNITED KINGDOM)

Manure	DM (%)	N_{tot} (kg ton^{-1})	P_{tot} (kg ton^{-1})	N_{tot}:P ratio	N_{RAN}:P ratio[1)]
Cattle FYM[2)]	25	6	1.5	4	1
Pig FYM	25	7	3.1	2.3	0.6
Cattle slurry	6	3	0.5	6	3
Pig slurry	4	4	0.9	4.4	2.7
Broiler litter	60	30	10.9	2.8	1.1
Layer manure	30	16	5.7	2.8	1.4

[1)] N_{RAN}, readily available N (NH_4^+-N in cattle and pig manures, and NH_4^+-N + uric acid N in poultry manures); [2)] FYM, farm-yard manure

An important advantage of integration of animal manures in the fertilization plan of crops, is the application of P, K and other secondary and micronutrients *via* the manure. In situations where only chemical fertilizers are used, supply of these nutrients is often insufficient. This has been mentioned as one of the possible causes of the yield decline observed in long-term experiments with rice in monoculture and rice-wheat rotations [6].

The use of a simple manure N Decision Support System (DSS) such as MANNER [26], is entirely consistent with the need for an integrated policy for the use of manures and mineral fertilizers.

7.5. MANURE APPLICATION AND THE ENVIRONMENTAL EMISSIONS

When nutrients from manure are applied in excess of crop requirements, there is a considerable risk of the surplus being lost to the wider environment. This is also true for poorly timed applications (autumn and winter in Western Europe) and for surface-spread slurries. Poor manure management may result in the ‘dumping’ of manure on land as a disposal operation and, as a result, possible direct discharge to surface waters. Moreover, because of a lack of awareness about the potential nutrient value from manures, the risk of diffuse pollution across a wide area is greatly increased [27].

There are two major pathways of nutrients loss from the applied manure to the wider environment (Figure 1.1). These are to the atmosphere and to the aquatic system. Of the nutrients from the applied manure, N is particularly important. Nitrogen is closely involved in crop production, but is also lost to the environment. Nitrogen is lost to the atmosphere *via* NH_3 volatilization and nitrous oxide (N_2O) emission. It is also lost to the aquatic systems, as NO_3^- leaching to ground water or NH_4^+ loss *via* surface run-off or by-pass flow to surface waters. Phosphorus loss *via* P leaching from P saturated soil may result from an excessive build-up of soil P status, following poor accounting for manure and chemical fertilizer applications.

7.5.1. Ammonia volatilization

Animal manure is a major source of atmospheric NH_3, contributing more than 50% to the global emission [28]. Ammonia is an important atmospheric pollutant with a wide variety of impacts. In the atmosphere, NH_3 neutralizes a large portion of the acids produced by oxides of sulfur (S) and N. A substantial proportion of atmospheric aerosols results from the chemical reaction of NH_3 with SO_x and NO_x, the products acting as cloud condensation nuclei. Much of the volatilized NH_3 is deposited in the vicinity of the source, *viz*. about 30%

within a radius of 5 km [29]. Ammonia deposition can contribute significantly to soil acidification, which is of particular concern in some woodland soils. It can also raise N levels in nutrient-poor soils and related botanically-rich habitats, for example old meadows and heath lands, thus changing the types of plants that grow there and reducing bio-diversity. Deposited NH_3 can also contribute to NO_3^- leaching losses.

Ammonia losses in agriculture occur primarily from the surface layers of NH_3-containing liquids, such as animal slurries and urine [30]. Ammonia can be volatilized rapidly into the atmosphere from animal slurry applied on the land, as well as from stores and livestock buildings as described in Chapter 5 in this document. The rate of NH_3 volatilization from slurry applied onto land usually reaches its maximum level immediately following the application (Figure 7.4). Then the rate decreases rapidly and NH_3 volatilization is almost complete within about 2–3 days following application. The volatilization could be described mathematically by a Michaelis-Menten type equation [31]. Ammonia volatilization is affected by many factors, including the concentration of NH_3 at the liquid surface, which is primarily a function of the chemical and physical conditions within the manure, in particular of total ammoniacal N (TAN = NH_4^+ + NH_3) content and pH. The transfer of NH_3 from the manure surface to the atmosphere is primarily a function of the local meteorological conditions. It is enhanced by factors stimulating evaporation, for example the area of the manure exposed to air, temperature, relative humidity and wind speed [32]. The TAN content of manure is related to animal feeding management (Chapter 4). The area of manure exposed to the air and the exposure time are related to the management of manure application [30].

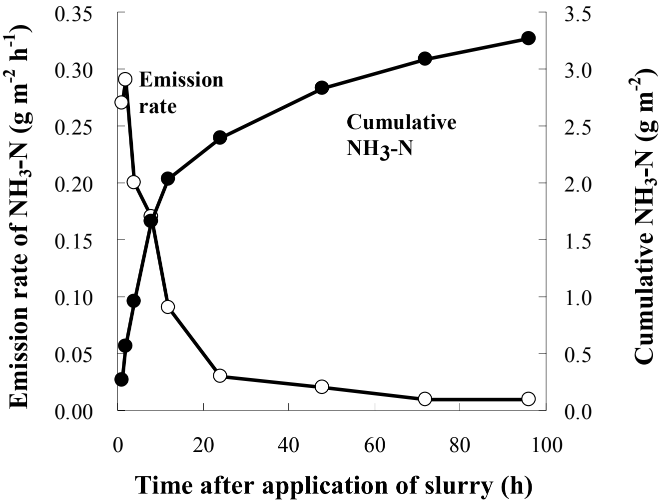

FIG. 7.4. Typical pattern of NH_3 emission and cumulative NH_3 emission with time, following surface application of cattle slurry to grassland [33]. Slurry application rate: 4 kg m^{-2} (9 g NH_4^+-N m^{-2}); pH of the slurry: 7.5.

Ammonia volatilization from animal manure is inevitable, following land application. In order to reduce these losses, however, some low-emission slurry application techniques have already been developed. The use of new machines such as trailing hose, trailing shoe (band-spread), and shallow injection significantly reduces NH_3 volatilization following application to grassland (Figure 7.5). For example, in recent research NH_3 volatilization was reduced with slurry applied by trailing hose, trailing shoe and shallow injection techniques by 26%, 57% and 73%, respectively, compared to the conventional splash-plate method [34].

Similarly, for cattle and pig slurry applied to arable land, the mean total NH_3 volatilization, expressed as % of NH_4^+-N applied, was 68% for surface spreading, 17% for surface incorporation and 2% for deep placement [19]. Of course, these new machines are considerably more expensive than the simple surface application techniques. However, by greatly reducing NH_3 loss, they make animal slurries more reliable N sources. In addition, the new techniques give a better distribution of manure over the field, and they tend to avoid excessive rates of application. The most effective methods for reducing NH_3 volatilization from solid manures involve incorporation of the manure; this should be undertaken immediately after the application, although emission rates are lower from solid manures than from slurries.

FIG. 7.5. New slurry application equipment for reduction of NH_3 emissions.

7.5.2. Nitrous oxide emission

Nitrous oxide (N_2O) is a potent greenhouse gas, constituting 6% of the anthropogenic greenhouse effect, and contributing to the depletion of stratospheric ozone. The global warming potential of N_2O for a time horizon of 100 years is estimated to be 296 times that of carbon dioxide [35]. Neither the sources nor the causes of the increase in N_2O of 0.7 ppb (parts per billion, volume basis) per year are well known. It is generally accepted, however, that soils are the most important source followed by the oceans, although there is uncertainty regarding the magnitude of the sources themselves. The anthropogenic sources that have been identified include: agricultural fields amended with N fertilizers and animal manure, animal manure stores, sewage, industry, automobiles, biomass burning, land clearing, and trash incineration. The contribution of agriculture to the global N_2O emission is about 35% [36].

The N_2O is produced through both nitrification under aerobic condition of the soil and denitrification under anaerobic soil conditions. Application of animal manure enhances denitrification and N_2O emission through supplying readily biodegradable organic matter and inorganic N, and by creating anaerobic conditions [37–39]. However, because of the extremely high spatial and temporal variability, considerable uncertainty persists in the N_2O emission from the soil receiving manure and N fertilizer. The N_2O flux following manure application is therefore often most readily detected just after the application, *e.g.* [40].

Soil water condition is one of the factors responsible for the uncertainty of N_2O emission from the soil. In general, microbial activity peaks at 30–60% of water filled pore space (WFPS). Nitrification and associated N_2O production also show maximum activity at 30–60% of WFPS, while optimum conditions for denitrification may occur at 60–80% of WFPS [41]. For the results presented in Figure 7.6, the soil water condition was roughly 70–80% of WFPS, whenever high N_2O emission fluxes were detected. This close correlation between high WFPS and high N_2O flux suggests that N_2O is often generated *via* the denitrification process. The low emission rates after July 2002 possibly indicate low contents of inorganic N in the soil.

Application time, rate and method of manure application also influence N_2O emission (Table 7.6). However, the effect of manure application method on N_2O emission from grassland was dependent on the application time. Shallow injection of slurry in March, when WFPS was high, resulted in significantly greater emissions of N_2O than surface broadcasting. In contrast, shallow injection in June appeared to reduce N_2O emissions. Doubling the rate of application increased N_2O emission 3.7-fold following surface application in November.

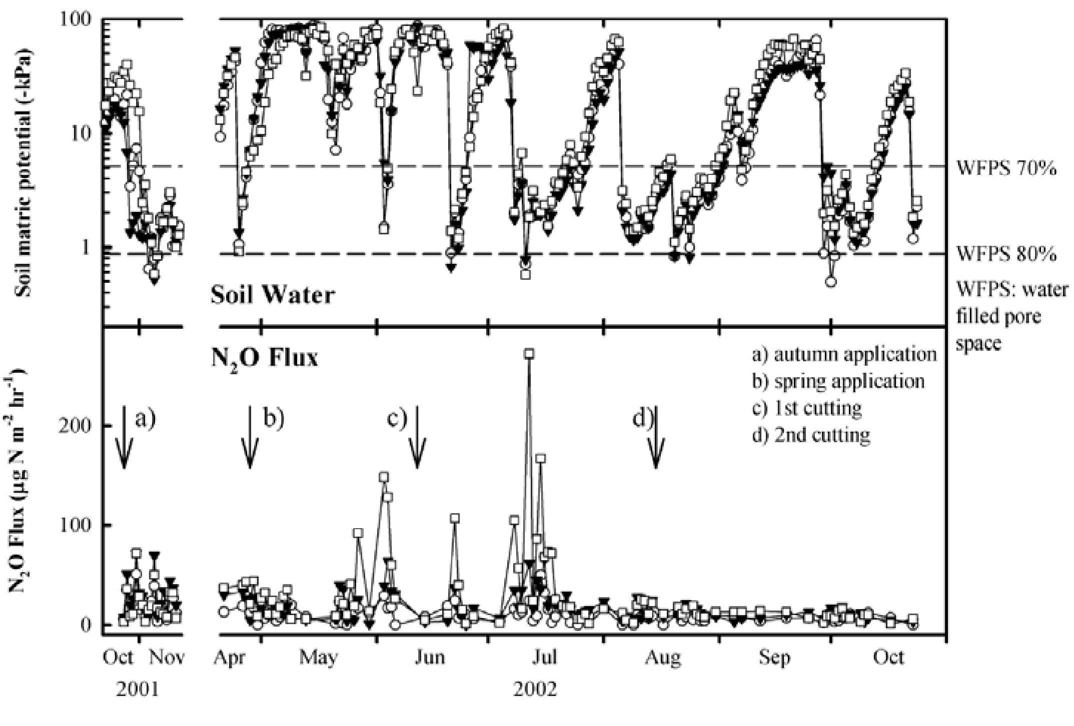

FIG. 7.6. Typical N_2O fluxes following the application of anaerobically digested cattle slurry to grassland [42].
○ Control; ▼ Heavy application (16 g m^{-2} as NH_4^+-N and 46.4 g m^{-2} as total N in the slurry) in autumn;
□ Heavy application (16 g m^{-2} as NH_4^+-N and 31.2 g m^{-2} as total N in the slurry) in the following spring.

TABLE 7.6. NITROUS OXIDE (N_2O) EMISSION FROM GRASSLAND AS AFFECTED BY PERIOD, METHOD AND RATE OF SLURRY APPLICATION [40]

	March (72 days)		June (89 days)		November (117 days)		
	25S	25I	25S	25I	50S	25S	25I
TAN applied, kg N ha^{-1}	44	47	32	32	56	28	28
Total N applied, kg N ha^{-1}	72	76	44	44	124	62	62
Net N_2O loss, kg N ha^{-1}	0.03^a	0.08^b	0.05	0.01	0.26^y	0.07^z	0.05^z
N_2O loss, % of TAN	0.07^a	0.17^b	0.15	0.03	0.47	0.24	0.19
N_2O loss, % of total N	0.04^a	0.10^b	0.11	0.02	0.21	0.11	0.08

TAN = total ammoniacal N of the slurry; 25S, 25I and 50S represent target rates and methods of application, *viz*. 25 m^3 ha^{-1} surface spreading, 25 m^3 ha^{-1} shallow injection, and 50 m^3 ha^{-1} surface spreading; $^{a, b, y, z}$, mean value with different letters are significantly different at P<0.05 level

In general, the emission factor of N_2O, expressed as a ratio of the emitted N_2O-N to the total N applied from the manure and/or chemical fertilizer, varied between 0.001 and 0.05. The Intergovernmental Panel on Climate Change (IPCC) recommended using a mean value of 0.0125, when the emission factors for different N inputs, *i.e*. N fertilizer, animal manure, crop residue, deposition and biological N fixation, are not well known [43].

7.5.3. Nitrate leaching

Environmental problems caused by manure N are also associated with the movement of NO_3^- through drainage waters to the ground- and surface-waters. It is commonly accepted that the limit of the NO_3^--N concentration in drinking water is 11.3 mg l^{-1}. For other purposes, like recreation, biodiversity and nature conservation, the Maximum Acceptable Concentrations of total N in water are lower. The leaching of NO_3^- to ground- and surface-water is one of the causes of eutrophication of aquatic systems.

The quantity of NO_3^--N lost in drainage water depends on the amount of residual NO_3^--N in the soil profile and the volume of water draining through the soil. Nitrate leaching can readily be understood by consideration of agronomic as well as hydrological aspects.

7.5.3.1. Agronomic aspects

Nitrate leaching depends on the difference between N supply (from natural sources, manure and fertilizer) and N uptake by the crop. On grassland, the rate of N application has been shown to determine leaching loss, rather than the source of N (manure or fertilizer) [44–46]. Furthermore, the period of application is important (*i.e.* matching of N supply and N uptake) and this has been shown in both arable experimental sites (Figure 7.7; [47]) and on grassland [22].

7.5.3.2. Hydrology

Precipitation and irrigation rates, along with soil texture and structure, influence leaching rates. Sandy soils in humid regions are particularly susceptible to NO_3^- leaching, while such nutrient loss in non-irrigated arid and semiarid soils is generally very low. Experimental data also show that the amount of excess winter rainfall is a major factor affecting the extent of NO_3^- leaching following manure application (Figure 7.7).

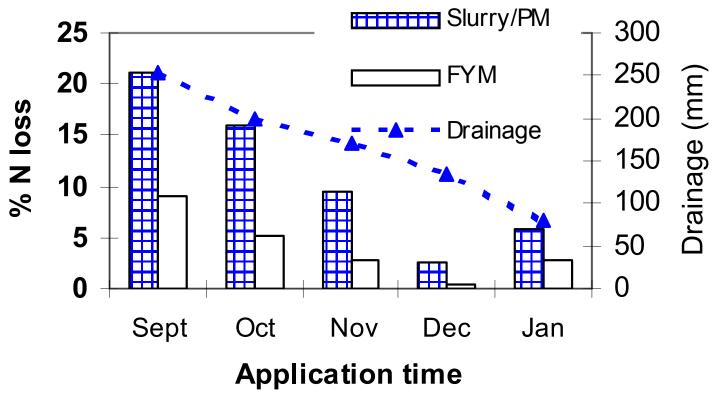

FIG. 7.7. Nitrogen leaching and estimated drainage following slurry, poultry manure (PM) and farm-yard manure (FYM) applications to freely draining arable soils (9 site years of data, 1990/91–1993/94; [47].

Manure type, through its content of inorganic N, is also an important factor affecting NO_3^- leaching loss. It has been demonstrated experimentally that N applied in slurry and poultry manure (with a large fraction of uric acid) in autumn and early winter is at much greater risk of leaching loss than N applied at the same time in FYM (Figure 7.7). This is simply a result of the much higher soluble or readily available N content in slurries and poultry manures compared with solid FYM, which often contains only ≤10% inorganic N.

Thus, the major causes of increased NO_3^- leaching from applications of manure have been shown to be due to incorrect timing (Figure 7.7) and excessive application rates [48]. For example, in the UK, animal manures are still commonly applied to soils in the autumn and/or at rates that supply N in excess of crop requirement [49]. Effective control, therefore, can be achieved by relatively simple adjustments in farming practice, such as the control of rates applied and the re-scheduling of manure applications, although the latter may require increased manure storage capacity.

7.5.4. Phosphorus leaching

Animal manures are an important component of the phosphorus (P) cycle in agriculture and their management influences the potential for P loss [50]. Following manure applications, heavy rains can induce run-off and erosion losses from the treated areas. Significant quantities of soluble and particulate P are thereby carried into surface waters, where eutrophication occurs. Phosphorus concentrations in run-off from heavily stocked catchments are, therefore, usually greater than those in lightly stocked catchments [51].

In intensive animal production areas, P losses can occur to ground and surface waters as a result of incidental losses in surface and sub-surface flow, directly from manure stores or following surface applications to fields or, indirectly following soil P enrichment. Surface run-off losses have been shown to depend not only on the rate and timing of manure application, but also, most importantly, on the time interval between the application and the run-off event [52–54].

In the UK, Smith and coworkers [21] studied P surface run-off losses following organic manure applications to land, utilizing a purpose-built facility on a 5° sloping site under arable tillage over a 4-year period. The application of cattle FYM and, especially slurry, to the silty clay loam soil increased both particulate and soluble P loss in surface water flow. Increased application of slurry solids increased all forms of P loss *via* surface run-off. Their results suggested that a threshold for greatly increased risk of P losses *via* this route, as for N, occurred at *ca*. 2.5–3.0 t ha^{-1} solids loading (Figure 7.8). This approximates to the 50 m^3 ha^{-1} application rate limit suggested for slurry within the UK 'good agricultural practice', though of course this limit also depends on slurry N and P content, in relation to crop demand. Although the losses recorded in this research were insignificant in agronomic terms (*viz*. < 2 kg P ha^{-1}), peak concentrations of P (up to 30,000 µg total P $litre^{-1}$) in surface water during a run-off event, could be of considerable concern in sensitive catchments. Losses of slurry P *via* surface run-off could make a significant contribution to accelerated eutrophication on entry to enclosed waters, particularly when combined with high concentrations of NO_3^--N.

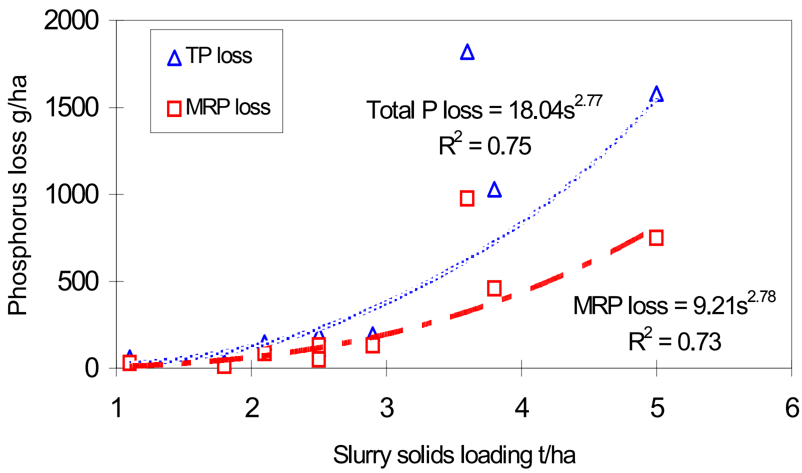

FIG. 7.8. Total losses of P in surface run-off from slurry solids loadings over 4 years of studies [21]. TP – total phosphorus; MRP – molybdate reactive phosphorus.

Accumulation of P in soils in regions with intensive animal production has also been a serious concern, for example in The Netherlands, Belgium and Northern Ireland. This must also be a concern in some Asian regions, where the intensity of animal production has increased rapidly. In these circumstances, the P accumulation increases the risk of P leaching losses [50] and the impact of particulate P associated with soil erosion. Moreover, too much accumulation of P in agricultural soils means a loss of a limited resource, because the global stocks of good-quality rock phosphates are rapidly decreasing and this should be a serious point of concern for future food supply.

Restricting manure application rates to those consistent with good agronomic practice, and within the limits specified in existing guidelines on good agricultural practice, offers the simplest and most effective control measure against this potentially important source of diffuse pollution.

7.6. FIELD APPLICATION OF MANURES

Knowledge of manure application rate is essential for the sustainable recycling of organic manures, correct fertilizer planning and reducing environmental pollution. Farms can improve the use of manure and slurry by applying them to land at known rates. Knowing the rate of manure application is the first step to improved control. The most important information is not difficult to obtain:

- the capacity of the slurry tanker or manure spreader or the pump used for irrigation;
- the field area, the number of loads applied and, hence, the application rate;
- the nutrient content (N, P and K).

A slurry tanker has a specified capacity, which will normally be filled during the spreading operation. This means that application rate can be estimated from the area spread (spread width x discharge length) and the capacity; and the average field rate, from the number of loads applied over the field area. Where slurry or effluent is delivered to the field *via* a pump and supply pipe to a sprinkler system or tractor-mounted applicator, the pump capacity and operating time will provide the delivery volume and the rate calculated from the area covered. Where the pump capacity is not known, this can be estimated by collecting the delivery in a tank of known capacity over a measured time, repeated several times for accuracy.

In the case of spreading solid manure, farmers are sometimes advised to weigh the manure spreader in order to assess the payload reliably [payload: the weight of manure carried by the spreader]. This usually requires weighing the spreader both full and empty, assessments which may be carried out at a farm or public weighbridge [55]. However, such advice is unlikely to be practical in Asian farming situations with limited, if any, access to either farm or commercial weighbridge facilities. In these circumstances, an estimate may be based upon:

(i) the volumetric capacity of the machine;
(ii) the bulk density of the manure.

There will usually be a manufacturers declared spreader volume or, otherwise, this can be estimated by measuring the internal dimensions (length, depth and width) of the spreader body. In a recent study in England [56], the bulk density of FYM averaged 0.7 t m^{-3}, and of poultry manure 0.5 t m^{-3}, while a range of 0.2 t m^{-3} to 0.5 t m^{-3} was reported elsewhere for poultry manures [57]. Probably, such variability depends substantially on the DM content of the manure. In practice, spreaders are often overloaded, above the top of the machine sides, and it is also necessary to take account of this machine ‘overload’ in assessing the total load volume (Figure 7.9 a). Thus, if the height of load above the machine sides were estimated at 0.5 m, for a spreader of 1.5 m depth and 7.5 m^3 volume, this would represent an increase in load volume of *ca*. 33%. Load volume would then be estimated at roughly 10 m^3, giving a total payload of 7 tonnes, assuming FYM of average bulk density. As in the case of the slurry tanker, application rate can then be estimated from the area applied (spreading width and discharge length), or record of the number of loads applied to the field of known area (Figure 7.9 b). These guidelines have been developed in relation to the mechanized handling and spreading of manures at the field scale on intensive farms, but the same principles should apply when dealing with relatively small amounts of manure on smallholder farms in parts of Asia. The manure may be handled in carts and spread by hand over plots of perhaps only a few m^2, however, it is suggested that estimation of area spread and load capacity will still provide the information needed to calculate rates of manure (and nutrients) applied.

(a) weighing of spreader using electronic pads; machine overfill height *ca*. 0.3 m.

(b) loads applied to the field should be recorded.

FIG. 7.9. Assessment of manure application rate should take account of (a) height of manure above the machine sides and (b) the number of loads applied to field area.

Current guidelines for the application of organic manures suggest a target spreading uniformity with a lateral coefficient of variation of not more than 25% [55]. However, it has been shown that precision in spread pattern can be considered of lesser importance than application rate [58]. Accurate information about rates of manure application is of crucial importance, without which it would be impossible to assess the rate of manure nutrients applied correctly. Failure to manage manure application correctly is likely to result in adverse impacts on crop yield and quality because of under- or over-application of nutrients, as well as potential for both point source and diffuse pollution. Many of the basic concerns of farmers about spreading imprecision can be overcome by simple adjustments and good machine set up. This requires selection of an agronomically sensible application rate and correct machine setting for the chosen rate. Slurry application equipment designed for abatement of NH_3 emissions, involving band application or shallow injection [34], also supplies the benefit of increased precision, both in terms of application rate and evenness of spreading pattern. However, such techniques hardly seem relevant to most Asian farming systems at the current time.

7.7. UTILIZATION OF MANURES – RECOMMENDATIONS

In this chapter, an attempt has been made to draw together general recommendations from the experiences of manure application systems developed in Europe, Japan and elsewhere, where these are thought to have relevance to livestock production in Asian countries. However, it is clear that there is an urgent need in these countries for R&D projects aiming at sustainable use of livestock manures in crop production. Important aspects are: design of proper manure handling and storage systems, manure sampling and analysis, development of manure application equipment, evaluation of crop responses and nutrient utilization and, based on that, development of fertilizing recommendations that integrate use of animal manure and chemical fertilizers. Some of the approaches that might be used within this research have also been proposed.

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8. CONCLUSIONS AND RECOMMENDATIONS

One of the key objectives of the International Atomic Energy Agency (IAEA) and the Regional Cooperative Agreement for Asia and the Pacific Region (RCA) project entitled "Integrated approach for improving livestock production using indigenous resources and conserving the environment" (RAS/5/044) is to improve the profitability of small holder farmers in the region by increasing productivity without damaging the environment. The counterparts of this project identified that one major opportunity to address this objective was through better manure management. They also recognised that there was an urgent need to change manure management practices because current practices are already damaging human, animal and environmental health and are not sustainable. Fifteen conclusions and recommendations were developed at the experts meeting on 'Development of guidelines for efficient manure management in Asian livestock production systems' held in Ho Chi Minh City, Vietnam, which are listed below. This comprehensive list of conclusions and recommendations covers aspects of manure management from social science issues, for example increasing awareness of the need for proper manure management and incentive schemes to encourage adoption, to complex biological issues related to integrated management practices and nutrient budgets.

The conclusions and recommendations are:

(1) There is an urgent need to raise the awareness of proper manure management. In particular, the possible effects on spreading of livestock diseases to humans, animals and crops and the impact of these diseases on human health, world trade and economy must be highlighted.

(2) Little information is available on the implications for disease transmission of different animal manure management practices. There is an urgent need for Member States in the Asia and Pacific Region to conduct research in this field.

(3) Legislation should be put in place to prevent discharge of animal manure (including liquids) to surface waters (irrigation and drainage channels, rivers, ponds and lakes).

(4) Considering the lack of a sustainable future for livestock production in urban environments, the development, or continuation, of livestock rearing in the vicinity of urban areas should be discouraged.

(5) Development and adaptation of technologies for effective manure management should be undertaken, as appropriate for Asian livestock production systems, and should be accompanied by the introduction of legislation for effective uptake of technologies.

(6) Surveys of current practices of manure application within crop production systems should be undertaken, and the effects of manure applications on crop yield and soil quality should be studied. The use of manure as a source of organic matter in poor and degraded soils should be given attention by farmers, extension personnel and those presons responsible for land management.

(7) Animal manure management lies at the interface between animal production, soil science and plant production. In order that animal manure is used effectively as a resource and, therefore, not a source of pollution, there is a need to enhance cooperation between scientists from these fields. A systems approach will encourage

the integration of animal, soil and plant components, thereby facilitating effective manure management.

(8) An integrated system for the use of organic manure and inorganic fertilizers offers the greatest potential for the sustainable management of manures while minimizing environmental pollution and developing confidence amongst farmers in the use of animal manures.

(9) The integrated use of animal manures, in combination with inorganic fertilizers, is likely to result in increases in crop and forage production, resulting in economic benefit for farmers, as well as environmental benefits.

(10) The development of pilot/demonstration farms (incorporating appropriate levels of technology) is felt to be an effective way of promoting positive messages on manure management at the national level.

(11) Development of decision support software can assist in manure management system design and in improved understanding of nutrient fluxes following manure application. This technology can be very effective in the promotion and uptake of sound manure management practices.

(12) Training courses on nutrient budgets and improved manure management practices should be designed and an initial test-run conducted for possible wider dissemination in Asian countries.

(13) It was felt that a website on manure management should be created under the Regional RCA Project of the IAEA.

(14) Based on current understanding and experience, a number of information sheets/pamphlets, on 'best practices' of manure management should be prepared and distributed to Asian countries, for use by extension workers and farmers.

(15) The initiation of programmes to reward farmers for carrying out sustainable manure management activities is felt to be a particularly useful strategy. This is believed to be an effective way of motivating other farmers to adopt 'best' manure management practices.

The first seven chapters in this document explain the principles and technical aspects of good and bad manure management practices, highlight the serious consequences of poor management and provide practical considerations for adopting new management practices. Presenting the 15 conclusions and recommendations in this final chapter of the guideline document is essential because they go beyond a simple review of existing practices and encompass a much broader checklist of what needs to be done by farmers for more sustainable manure management in the region. This checklist should be used as the framework for directing all activities that Member States in the region must initiate to ensure the long term success of the programme.

It is evident from the literature reviewed that little basic information on nutrient transport and budget is available for Asian livestock production systems. Various research options have been highlighted in this document. Nuclear techniques play an important role in generating quantitative data on farm nutrient balance, nutrient availability to crops from manure, effect of animal diet on nutrient release in manure, etc. The techniques based on stable isotopes, 15N and 13C have comparative advantage over the conventional techniques because of their high sensitivity and specificity, enabling generation of reliable data. A few examples of the use of isotopic approaches are the use of natural abundance levels of nitrogen to determine the manner in which the flow of nitrogen at various physical scales is controlled in an ecosystem. The areas within crop-livestock systems, requiring further definition to enhance nitrogen utilization, which can be evaluated by 15N studies are: the effect of quality of diet on nitrogen utilization and partitioning into faeces and urine; the dynamics of nitrogen turnover from faeces and urine, plant residues and soil organic matter and the impact of changes in husbandry and management practices; spatial and temporal effects of excretal return (application after storage or at grazing); interactions between nitrogen, other nutrients and water availability; nitrogen sources and rates of transformation and transfers into loss pathways and construction of system nutrient balances; and identification and determination of uptake rates of nitrogen by plants from soil, fertilizer, manure or atmosphere. In livestock research, several studies have used 15N enriched plant material fed to animals to generate 15N-labelled excreta for research on the fate of excreta N, and for obtaining better understand of the variability of nitrogen supply from manure in relation to feed quality. Similarly, foliar 15N labelling has been used to better quantify root N yields and to determine the uptake of 15N labelled root N by subsequent crops. In addition, 34S could be used to construct sulphur budgets and to follow pathways of sulphur in the soil/plant/animal continuum. 32P or 33P is used to estimate the efficiency of P utilization in leaf production in legumes used for livestock feeding. In developing countries, there is a widespread occurrence of P deficiency and P fertilization enhances crop biomass production and quality, which when fed to livestock could affect manure quality. In order to study the primary and interaction effects of nutrients in the soil/crop/animal continuum, multi-labelled plant material could provide valuable information. The data from such studies is vital for developing practical recommendation systems for optimum use of manure in cropping systems.

LIST OF PARTICIPANTS

Boettcher, P.J.	Joint FAO/IAEA Division of Nuclear Techniques in Food and Agriculture, Department of Nuclear Sciences and Applications, International Atomic Energy Agency, Wagramerstrasse 5, P.O. Box 100, A-1400 Vienna, Austria p.j.boettcher@iaea.org
Makkar, H.P.S.	Institute for Animal Production in the Tropics and Subtropics (480b), University of Hohenheim, 70599 Stuttgart, Germany Email: makkar@uni-hohenheim.de
Martinez, J.	La Recherche pour l'Ingenierie de l'Agriculture et de l'Environnement, 17, avenue de Cucillé, 35044 Rennes, France Email: Jose.Martinez@cemagref.fr
Matsunaka, T.	Faculty of Dairy Science, Rakuno Gakuen University, Bunkyodai-Midorimachi 582-1, Ebetsu, Hokkaido 069-8501, Japan Email: matsunaka@rakuno.ac.jp
Ong, H.K.	Malaysian Agricultural Research and Development Institute(MARDI), Strategic Resources Research Centre, Pejabat Pos Besar, P.O. Box 12301, GPO 50774 Kuala Lumpur, Malaysia Email: keng@mardi.my
Ramat, I.	Department of Agriculture, Bureau of Agricultural Research, Elliptical Road, Corner Visayas Avenue, Quezon City, Philippines Email: ibramat2005@yahoo.com

Smith, K.	ADAS Laboratories Wolverhampton, Wergs Road, Woodthorne, Wolverhampton, WV6 8TQ Staffordshire, United Kingdom.K. Email: ken.smith@adas.co.uk
Sommer, S.G.	Danish Institute of Agricultural Sciences (DIAS), Research Centre Bygholm P.O. Box 536, 8700 Horsens, Denmark Email: SvenG.Sommer@agrsci.dk
Van der Meer, H.G.	Plant Research International, Wageningen UR, P.O. Box 16, 6700 AA Wageningen Building No. 122, Bornsesteeg 65 6708 PB Wageningen Netherlands Email: hugo.vandermeer@wur.nl
Vercoe, P.E.	The University of Western Australia, 35 Stirling Highway, Crawley WA 6009, Australia Email: pvercoe@cyllene.uwa.edu.au