Nutrient imbalances in China with the focus on N and P in agricultural systems in Jiangsu Province

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Abstract

During recent decades, the use of mineral fertilisers, especially nitrogen (N), has increased significantly in China compared with the use of organic fertilisers, such as crop residues and human, animal and green manure. At the same time, an ever-increasing population is putting pressure on the agricultural sector due to their increased demand for food and is competing for arable land due to a rapid, ongoing urbanisation and industrial expansion. China now faces serious environmental problems, such as eutrophication and degradation of drinking water quality due to heavy loads of polluted, untreated wastewater from households and industry and losses of excessive fertiliser N from farmland. This report reviews the nutrient imbalances in Chinese agriculture, especially in Jiangsu Province. The process-based simulation model GLEAMS was used to estimate the N and phosphorous (P) losses and transformations in an intensive vegetable-producing peri-urban area in Shiba Village, Nanjing City, Jiangsu Province. The results showed that currently, there is an enormous surplus of N in the system, resulting in high losses of N by leaching and denitrification and very low nutrient use efficiency. According to the simulation results, the nutrient use efficiency may be improved and the losses of N significantly reduced by decreasing the fertiliser input to twice the nutrient yield in the harvest, without any loss in crop yield.

Nutrient imbalances in China with the focus on N and P in agricultural systems in Jiangsu Province – a literature review

Introduction

China is the third largest country in the world in terms of land area with its 9 571 300 km^2. In 1990, the country had the highest population in the world, 1 133 683 000 persons (1.1 billions), of which 32% lived in cities. In 1999, the excess of births over deaths was 8.7 per 1000 capita (Statistics Sweden, 2003). Even if this number has decreased since 1993, when it was 11.3 (Statistics Sweden, 1998), the rapidly increasing population is putting pressure on the agricultural sector through an increased demand for food. It is estimated that the population of China will have increased to 1.6 billions by 2030 and that 0.7 billion tons of grain for food will be needed each year (Jin and Jiang, 2002). The total yield of grain crops (wheat, maize, rye, barley, oats and rice) was about 0.4 billion tons in both 1996 (Statistics Sweden, 1998) and 2002 (Statistics Sweden, 2003), which means that the yields must nearly double until 2030.

China is also the world's largest fertiliser user (Li and Lin, 2000) and largest chemical N (nitrogen) fertiliser producer (Statistics Sweden, 1998) and has the largest domestic livestock population (Li and Lin, 2000). Since the late 1970s, a rapid shift from a labour-intensive, complex farming system based on recycling of nutrients to a system heavily dependent on mineral fertiliser, especially N inputs, and an increasing abandonment of organic and green manure has taken place in China (Richter and

Roelcke, 2000). An increase in cereal production in Asia between the years 1960 and 1997 (176 and 276 kg/person, respectively), is reported by Galloway (2000). The increase is primarily explained by an extensive use of commercial fertilisers (Galloway, 2000). According to Zhu and Chen (2002), there is a significant positive linear correlation between annual food production and annual fertiliser use throughout the period 1949-1998. Mosier and Zhu (2000) found that the relative grain yield per unit of fertiliser N input declined precipitously from 1961 until 1980, a period in which fertiliser consumption increased by a factor of 20 while grain production only doubled. As no further increase in grain yield was observed with increased fertiliser input after 1980, the authors suggest that on average in Asia, current increases in N input are no longer translated into increased grain yields. Meanwhile, environmental pollution due to nutrient losses has become a serious issue. For example, N concentration in groundwater and surface waters has increased and emissions of gaseous N from farmland are increasing (Zhu and Chen, 2002). The former contributes to eutrophication of lakes (Sun and Zhang, 2000) and thus, drinking water shortages and algae bloom (Zhu and Chen, 2002).

In 1995, 10.3% of China's land area was used for agronomic purposes (Statistics Sweden, 1998). This figure increased to 15.4% of the total in 2001 (Statistics Sweden, 2003), but the yields of grain crop did not increase at all, as described above (Statistics Sweden, 1998, 2003). In 1998, the arable land area per capita in China was lower than the world average, 0.1 and 0.23 ha/capita, respectively. Expansion of economic crop cultivation, industrialisation and urbanisation is now creating competition for the limited arable land resource, with the production of food needed to meet the demand of an ever increasing population. It is suggested that the only option to increase the food production relies on the further increase in yields. Increased amounts of fertiliser inputs are one common approach to achieve it (Zhu and Chen, 2002). Jin and Jiang (2002) suggest that the key to a sustainable development of food production in China lies in management of soil nutrients and an efficient allocation of inorganic and organic resources of nutrients. Richter and Roelcke (2000), studying the N-cycle in intensive agriculture, came to the conclusion that in a medium-term perspective, there seems to be little hope of reducing fertiliser use in China because of the enormous pressure on grain production caused by the population growth, changing food consumption patterns (increased consumption of animal products) and loss of agricultural land. According to Sheldrick et al. (2003), it is particularly important that China manage to achieve a correct balance of nutrient application. Unless this can be achieved, food production will be constrained and nutrient losses will continue to occur.

In this literature review, I attempt to describe some of the nutrient imbalances caused by agriculture in China and the environmental issues connected with these. I focus on Jiangsu Province, especially the area near Nanjing City, 300 km west of Shanghai, in order to prepare for a minor field study on nutrient management in Shiba Village, Maqun Town, a peri-urban area of Nanjing.

Nutrient cycling: N and P

An appropriate understanding of the nutrient imbalances in agroecosystems requires some knowledge about nutrient cycling in the soil system and in the large scale. Therefore, I begin this literature review with a description of the processes involved in the nitrogen (N) and phosphorous (P) cycles, in which the pools and flows of N and P are identified and those of major interest from an agricultural/horticultural point of view are explained more thoroughly.

The N-cycle

The N-cycle in soils is illustrated in Figure 1.

There are two main pools of N in the soil, namely N in living and dead organic matter, which is referred to as organic N and makes up 95-99% of the total soil content of N; and mineral N, such as NH_4^+, NO_2^- and NO_3^-. When talking about the N-cycle, there are some terms that are commonly used, summarised in Table 1. *Ammonification* is the release of NH_4^+ from organic compounds. The conversion of organic N into mineral N-forms (NH_4^+ and NO_3^-) is termed *mineralisation.* It is a series of reactions in which soil-dwelling microbes hydrolyse the organic N-compounds by enzymatic processes. The opposite of mineralisation is *immobilisation,* the conversion of mineral N into organic forms. *Nitrification* is the conversion of NH_4^+ via NO_2^- to NO_3^-. In the first step, the autotrophic bacteria *Nitrosomonas* enzymatically oxidises NH_4^+ to NO_2^-, and in the second, another autotrophic bacteria, *Nitrosobacter*, converts the NO_2^- to NO_3^-. The nitrification process can be slowed down or inhibited by the use of chemicals, called nitrification inhibitors. Both the mineralisation and nitrification processes release energy, while the immobilisation process retains it (Brady and Weil, 2002).

Denitrification is the term for a series of biochemical reduction processes that convert NO_3^- into gaseous forms of N that are then lost to the atmosphere. These transformations are also carried out with the help of microorganisms, mostly facultative anaerobic bacteria such as *Bacillus, Nitrosomonas, Micrococcus* and *Achromobacter.* The forms of gaseous N produced, which are NO, N_2O and N_2, depend on the oxygen content in the soil: the higher the concentration, the more oxygen in the gaseous N produced. For the denitrification process to take place, NO_3^- must be available but the soil oxygen content should not be too high (<10% in the soil air or <0.2 mg O_2/L in the soil solution), readily decomposable organic compounds that can provide the energy needed for the transformation must be available and the temperature and soil pH must be within the tolerance range of the bacteria (Brady and Weil, 2002).

Another way that N can be lost to the atmosphere is by *volatilisation* of NH_3. This transformation is completely chemical in its nature, explained by the equilibrium between $NH_3(g)$ and $NH_4^+(aq)$. The source of NH_4^+ may be manure, fertilisers, decomposing plant residues or the foliage of living plants. The NH_4^+ may also be adsorbed and fixed by clay minerals, as the ions are attracted to the negatively charged surfaces on clay and humus, either in exchangeable form or entrapped within the structure of certain clay particles. The latter is termed *fixation* (Brady and Weil, 2002).

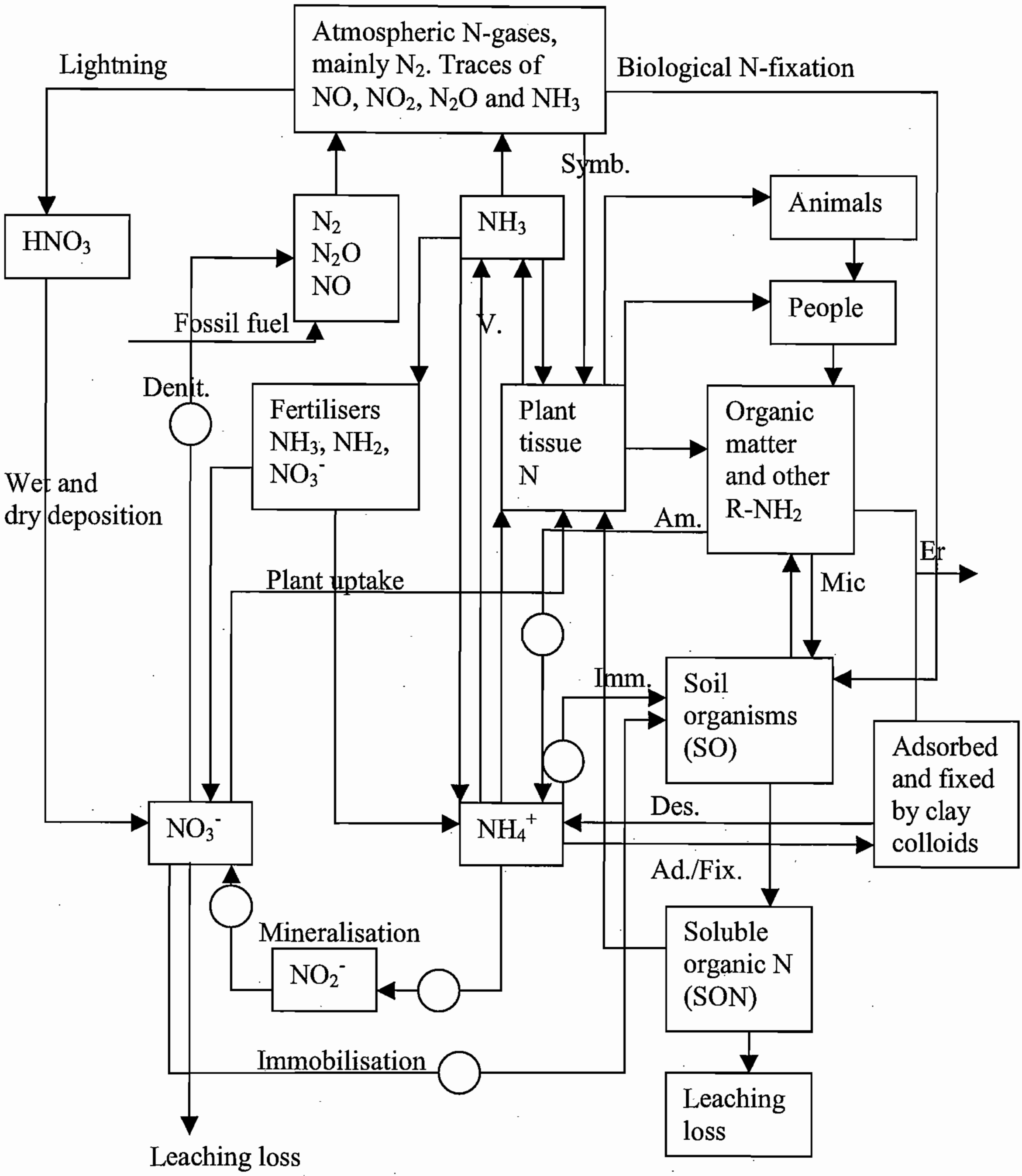

Figure 1. The N-cycle in soils. Boxes represent pools of N, arrows the transformations among these pools. The circles indicate transformations in which soil organisms are involved. Ad./Fix. = Adsorption or fixation, Am. = Ammonification, Denit. = Denitrification, Des. = Desorption, Er.= Erosion loss and runoff, Fossil fuel = combustion of fossil fuels, Mic. = Microbial synthesis, Symb. = Symbiotic biological N-fixation, V= Volatilisation. Adapted from Brady and Weil (2002).

Finally, the terms *symbiotic* and *biological N-fixation* refer to the conversion of atmospheric N_2 into organic N-compounds by certain bacteria, either in symbiosis with leguminous or other plants (*Rhizobium, Frankia*) or free living (*Azotobacter,* *Beijerinckia, Clostridium*). The most effective N-fixation is generally achieved by symbiotic associations between bacteria and leguminous plants, which may fix up to 280 kg N/ha and year. However, high levels of mineral N in the soil inhibit the N-fixation, for example if the application rate of fertiliser is high. The non-symbiotic N-fixation may supply 5-20 kg N/ha and year to the ecosystem (Brady and Weil, 2002).

Table 1. Processes involved in the N cycle in soils

Process	Driving force	Chemical reaction
Ammonification	Soil organisms	Organic N $\Rightarrow NH_4^+$
Biological N-fixation	Free living bacteria	N_2 -$\Rightarrow$ organic N
Denitrification	Microorganisms (bacteria)	$NO_3^- \Rightarrow NO_2$, NO, N_2O
Fixation	Attraction between positive and negative charges	NH_4^+ fixed by clay colloids
Immobilisation	Microorganisms	NO_3^- and $NH_4^+ \Rightarrow$ organic N
Mineralisation	Microorganisms	Organic N $\Rightarrow NO_3^-$ and NH_4^+
Nitrification	Autotrophic bacteria	$NH_4^+ \Rightarrow NO_3^-$
Symbiotic N-fixation	Bacteria in symbiosis with plants	$N_2 \Rightarrow$ organic N
Volatilisation	Chemical equilibrium reaction	NH_4^+ (aq) $\Rightarrow NH_3$ (g)

To summarise the fate of an NH_4^+-ion in the N-cycle, there are five different possibilities. First, it may be immobilised by microorganisms. Second, it may be removed from the soil system by plant uptake. Thirdly, the NH_4^+-ion may be fixed in the interlayers in certain clays of 2:1-type. The fourth pathway is the transformation to NH_3(g) and volatilisation to the atmosphere, while in the fifth, the NH_4^+-ion may be oxidised to NO_2^- and subsequently to NO_3^- by microbial action (nitrification). For a NO_3^--ion, there are four possible pathways. As in the case of the NH_4^+-ion, the NO_3^--ion may be immobilised by microorganisms or removed by plant uptake. However the NO_3^--ion may also be leached in drainage water or lost to the atmosphere by denitrification, producing several N-containing gases (Brady and Weil, 2002).

According to Mosier and Zhu (2000), the anthropogenic sources of N in agricultural systems are chemical fertilisers, animal manure, enhanced biological N-fixation by cultivation of N-fixing crop, recycled crop residues and cultivation of mineral and organic soils, which leads to enhanced mineralisation of organic matter. The N not utilised by the crop is either stored in the soil profile or is lost to the surroundings by leaching of NO_3^- to groundwater, runoff of soil or NO_3^- to surface waters, volatilised as NH_3 or lost by nitrification or denitrification.

It is estimated that the leaching losses of chemical fertiliser N applied to farmland are 0-19%. In field investigations in a double cropping system (wheat-rice) in Jiangsu Province, the total leaching loss was 1.8 and 3.4% of the chemical N fertiliser applied for the wheat and rice growth period, respectively. Another survey conducted in the same province found that the annual leaching loss from a wheat-rice rotation was 2.5-6.1% of the applied N (fertilisers and manure), with a mean value of 3%. The Chinese national averages of leaching and runoff losses of chemical fertiliser N are suggested to be equivalent to 2 and 5%, respectively, of the N applied (Zhu and Chen, 2002).

The P cycle

The cycling of P in soils is shown in Figure 2. .

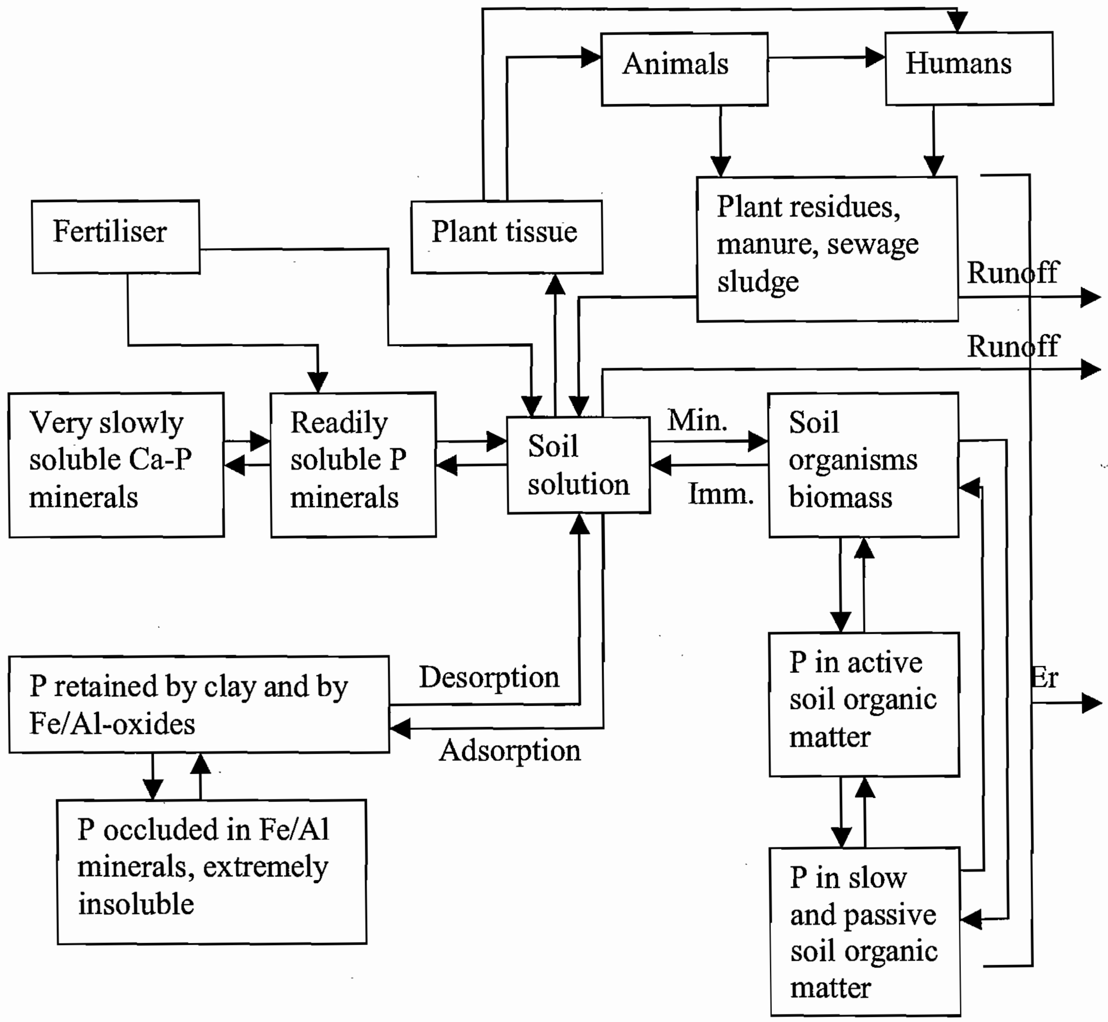

Figure 2. Phosphorus cycle in soils. Boxes symbolise pools of P in the cycles, while arrows indicate flows and transformations among these pools. Er. = loss on eroded particles, Imm. = immobilisation, Min = mineralisation, Runoff = loss in runoff. Adapted from Brady and Weil (2002).

There are three principal groups of P-containing compounds in soils. The first is the organic group of P-containing compounds, accounting for 20-80% of the total P content in surface soils. The second group is the calcium-bound inorganic P. It is the dominant form of inorganic P in high pH and calcareous soils, while in low pH and/or highly weathered soils, the predominant form is the iron- or aluminium-bound inorganic P, the third group. The amount of plant available P, which is in the soil solution, is usually very small, seldom exceeding 0.01% of the total soil P content. The plant root uptake of P can be significantly enhanced by mycorrhizae. By microbial immobilisation and adsorption of P on clay and Fe-/Al-oxides, P can be withdrawn from the soil solution, and is thereby not available for plant uptake. However, by mineralisation of organic matter and by desorption of adsorbed P, P can be released to the soil solution again. By application of readily soluble P in chemical fertilisers, the P-concentration in the soil solution can be increased rapidly, thus increasing the plant available pool. However, the enhanced availability is only temporary, since the P reacts fairly rapidly to form low-solubility compounds (Brady and Weil, 2002).

According to Brady and Weil (2002), P is lost from the soil system through plant removal (5-50 kg/ha and year in harvested biomass), erosion of P-carrying soil particles (0.1-10 kg/ha and year on organic and mineral particles) and in dissolved form in surface runoff water (0.01-3.0 kg/ha and year). Unlike N, there are no gaseous losses of P. The input of P by deposition is very low (0.05-0.5 kg/ha and year) (Brady and Weil, 2002).

Fertiliser use in China

Beginning with the green revolution in the late 1940s, a great achievement in food production has occurred in China. In 1949, the total food production in China was 113 million tons, corresponding to 1.0 ton/ha (Zhu and Chen, 2002). Almost 50 years later, in 1998, it had increased to a total of 512 tons, or 4.5 tons/ha. The increase is attributed to expanded cereal production, introduction of high yielding varieties, intensification of cropping (two or three seasons per year) and increased use of irrigation and fertilisation. Since the 1970s the consumption of chemical fertiliser N has also increased rapidly, from 2.865 Tg in 1970 to 24.8 Tg in 1998. The 1998 consumption in China is estimated to correspond to an annual N application rate of 180 kg/ha, with 120 kg/ha per cropping season (cropping index 1.5) (Zhu and Chen, 2002). According to Sheldrick et al. (2003), there is an imbalance between the fertiliser use among the Chinese provinces. The fertiliser application rates are high in high yielding regions, particularly in the coastal regions in the east, while some remote inland areas suffer from N depletion. However, as reported by Mosier and Zhu (2000) and mentioned in the introduction, the current average increase in N fertiliser in Asia is no longer thought to be translated into increased grain yields. According to Mosier and Zhu (2000), the estimated apparent efficiencies of chemical fertiliser N have also declined since 1961. With the period 1961-1965 as a reference, the efficiency had been halved until 1991-1993. In a survey in a double cropping system (maize-wheat) in the northern part of Jiangsu Province, a high content of mineral N was found in the upper 70 cm of a sandy loam alluvial soil after harvest of wheat and prior to maize sowing. This may be the result of a long period of continuous luxury application of fertiliser N rather than N surplus from the previous cropping season. In addition, estimations of total N input, output and balance showed a considerable surplus of N in the national balance sheet when the losses of chemical fertilisers, organic manure, soil N, leaching and runoff were taken into account (Zhu and Chen, 2002).

Liu et al. (2003), referring to the National Compilation Committee of Environmental Almanac 1996, compared the population, crop production, cultivated area and fertiliser use in the Changjiang River drainage basin in 1980 and 1996 (Table 2). During this period, the fertiliser consumption tripled, while the crop production increased by a factor of 1.5. While there was a population increase of 70 million people, there was a reduction of 17 million ha in the cultivated area.

Table 2. Population, crop production, cultivated area and fertiliser consumption in the Changjiang River drainage basin in 1980 and 1991 (Liu et al., 2003)

Year	Population	Crop production	Cultivated area	Fertiliser use
1980	340x10^6	112x10^6 tons	358x10^6 ha	3.2x10^6 tons
1996	410x10^6	170 x10^6 tons	341 x10^6 ha	9.67 x10^6 tons

Mosier and Zhu (2000) calculated the sources and amounts of fertiliser N inputs in Asia in 1961 and 1994 (Table 3). As can be seen from the table, the use of both chemical and all four organic sources of N increased during the period, but the relative importance of chemical fertilisers increased drastically compared to the other four sources.

Table 3. The sources and amounts of fertiliser N consumption in Asia in 1961 and 1994 (Mosier and Zhu, 2000)

Year	Unit	Chemical fertilisers	Animal manure	N-fixation	Recycling of crop residues	Total nutrient input
1961	Tg N	2.1	6.9	1.7	5	15.7
	% of total input	13.5	43.9			
	kg N/ha	6.7				
1994	Tg N	40.2	14.2	2.5	12.6	69.5
	% of total input	57.8	20.9			
	kg N/ha	108				

In the Taihu region, just at the junction of the northern and central subtropics in the southern part of Jiangsu Province, the agricultural practice is characterised by a very intensive double-cropping system. In the summers, irrigated rice is grown, while in the winters, upland winter wheat or rapeseed are the predominant crops. Application of organic manure, such as waterlogged compost, has almost been abandoned in favour of mineral fertilisers and winter green manures have been replaced by cash crops in the crop rotation. The average annual surplus of N is 217-335 kg/ha, and because of high transformation losses of N due to alternating water regime, the N utilisation rate by the crops is low (28-41%). Imbalanced N fertilisation is also reported from Northern China. In intensive vegetable production systems, 50% of 110 investigated locations had NO_3^- contents exceeding the critical WHO value of 11.3 mg NO_3^--N/L for drinking water in groundwater and drinking waters (Richter and Roelcke, 2000).

According to Zhu and Chen (2000), the use of organic N, such as human and animal excreta and crop residues, has recently declined. In 1998, only 33% of these resources was used, with the majority of the unused part released to the environment. The situation is particularly alarming in eastern China. A survey on rice fields in Jiangsu Province revealed that the application rate of organic manure to flooded rice had declined from 50 kg N/ha and year, 30% of the total N applied in 1986, to only 20 kg N/ha and year, or 6.7% of the total N applied in 1997. Human excreta from the urban population was used as fertilisers in the past, but is now mostly released to surface water. The use of pig and cattle manure in agriculture has also decreased (Zhu and Chen, 2000).

There are some general problems connected with the use of chemical N fertilisers. First, the N application rate is too high, especially in high-yielding regions and farmlands, which leads to low nutrient use efficiency and large losses (Zhu and Chen, 2003). Even if the use of N is regarded as balanced at the national level, the uneven distribution of fertilisers between regions causes environmental problems due to N surplus in some regions, while others suffer from N deficiency (Sheldrick et al., 2003). Secondly, improper application methods are widely used, such as broadcasting of NH_3-forming fertilisers, resulting in increased losses by volatilisation. Thirdly, a large portion of the fertiliser N is applied too early, when the uptake capacity of the crop is limited, and the fourth problem is an unbalanced N:P:K ratio (Zhu and Chen, 2003).

In 1997, the annual net depletion rate of P in China was 0.6 million tons, which was the same level as in 1961. From 1961, however, the annual P depletion rate increased steadily to reach 1.2 million tons/year in 1985. Due to an increased use of P fertilisers and manure, the trend of increasing P depletion rate has now been corrected according to the results from 1997 (Sheldrick et al, 2003).

Sheldrick et al. (2003) have used models to calculate nutrient balances (fertiliser+manure-nutrient in the crops) in China at a national and regional level. The results showed that most soils in China suffer from increasing potassium (K) depletion rates, since K inputs from fertilisers and recycled crop residues were relatively small compared to the output in the harvest. Most of the K was supplied by the soil K reserves, comprising 41% of the total K input, and most of the output K was found in the crop residues. The authors' conclusion was that China's future food security is seriously threatened by the high soil K depletion rate found in their study, increasing from 28 kg/ha in 1961 to 62 kg/ha in 1991. To reverse the trend and achieve K balance, thus preventing further increases in the K depletion rate, it is necessary to increase K fertiliser use and to improve the management of crop residues (Sheldrick et al., 2003).

In Jiangsu province, the K depletion rate in 1996, 101 kg/ha, was estimated to be even higher than the mean value for China, assuming that excreta production is proportional to the weight of the meat produced and that the area of arable land can be increased in proportion to the total increase in arable land in China. With this high rate of K depletion, even soils with high levels of exchangeable K were expected to be seriously affected in a medium-term perspective (Sheldrick et al., 2003).

Environmental impacts of imbalanced nutrient management

Imbalanced nutrient management pollutes the waters in rivers and lakes

Eutrophication has been identified as a major cause of impaired water quality and restricts the possibilities of water use for fisheries, recreation, industry and drinking. In the end, eutrophication results in oxygen shortages due to decomposition of algae and aquatic weeds, whose growth is accelerated by the increased nutrient supply to the water. Nitrogen, P and C are considered as essential for the growth of aquatic biota. For an improved nutrient management, P has been in focus since it is difficult to control the N-fixation by some blue-green bacteria and the N and C exchange between the atmosphere and water (Sharpley et al., 2000). In addition, it is relatively easy to remove the P from the water by waste water treatment (K. Blombäck, pers.comm.). Liu et al. (2003) found that in the Changjiang River, the third largest river in the world, the biological activity was strongly P-limited when the ratios of dissolved nutrients were compared with typical values for uptake of phytoplankton.

P from agricultural sources can represent a significant input to freshwater and if P accumulates in the soil, the amount of P in the drainage water increases (Sharpley et al., 2000). In the survey of Liu et al. (2003), no such increase in P concentration in the river waters was obvious between 1980 and 1996. However, the NO_3^- concentration in the river increased. During this period, the population increased by 20% and crop production and fertiliser use increased by 150% and 300% respectively, while the cultivated area decreased slightly. Calculations showed that the origin of the nutrients in the Changjiang River was the intensive agriculture and domestic production, reflected by lower nutrient concentrations in the upper reaches, where the population density is lower than in the south. In the south, the tributaries to the river are characterised by intensive anthropogenic perturbation. After the 'Three Gorges Dam' is completed, the nutrient levels are expected to increase further due to leaching from inundated soils and intensive human activity. Thus, the building of the dam will probably exercising a great deal of influence on the ecological environment of the Changjiang estuary and its adjacent sea (Liu et al., 2003). For example, algae bloom frequently occurs in some Chinese lakes (Zhu and Chen, 2002) and coastal waters (Duan et al., 2000), an indication that eutrophication is becoming a serious problem (Zhu and Chen, 2002).

According to Duan et al. (2000), the annual transport of dissolved inorganic nitrogen in the Changjiang River increased four-fold in a 29 year-period (1962-1990), with the largest increase during the 1980s. Nitrate contributed 86% of the transport, mainly occurring during periods of high flow in May to October with a peak in June. A positive relationship between the annual transport of dissolved inorganic nitrogen and the application of chemical fertilisers in the Changjiang catchment was recognised. Liu et al. (2003) found that the nutrient levels in the Changjiang River were comparable to those in European and North American polluted and eutrophied rivers, such as the Loire, Po, Rhine and Seine. However, chemical fertilisers are not the only source of inorganic nitrogen in Changjiang River. Inadequate disposal of industrial and domestic waste water will probably contribute to the ascending trend in nitrogen transport in the future, and during the ten years between 1980 and 1990, the amounts of wastewater in the Changjiang drainage basin increased from 127×10^8 m^3 to 142×10^8 m^3 (Duan et al., 2000). Cheng et al. (1996) found high amounts of organic and inorganic pollutants in monosodium glutamate wastewater provided by three mills in Jiangsu Province. The water from one of the mills had to be diluted 71434-fold before the wastewater could be safely discharged without being lethal to the fish *Ctenopharyngodon idellus*, which was used for a bioassay of the acute toxicity of the untreated water.

Sun and Zhang (2000) identified four sources of N in lake sediments, which directly reflect the rate of the lake's eutrophication process since the N content in the lakes is closely connected with the rate and level of eutrophication. The four sources of depositional N in lake sediments were: terrestrial from erosion from basin; pollution from industry and wastewater of urbanisation; autochtonous plankton and algae in the lake; and chemical deposition in a drought zone, where evaporation rates are high (Sun and Zhang, 2000). Research has shown that 7-35% of the total N load to water bodies originates from agricultural land (Zhu and Chen, 2002). In particular, lakes located near cities suffer from serious pollution by excessive N, for example the Xianwu Lake near Nanjing City (Sun and Zhang, 2000). Zhu and Chen (2002) suggest that the major contribution of N in waters might come from human and animal excreta rather than from soils. The suggestion is based on the fact that most of the civil sewage sludge and wastewater from animal husbandry in rural areas is released to the water without any treatment.

A historical review (Sun and Zhang, 2000) showed that the serious pollution resulting in eutrophication of the Chinese lakes developed during the 1980s, coinciding with policy expansion in China and a rapid industrial development. The authors mention Taihu Lake (southern Jiangsu) in the East China Plain lake region as an example. In this area, the lakes are shallow and connected with rivers, whose water flow contributes to a higher capacity of water alternation. Thus, the lakes in this region have lower N contents than the rivers in the northern and the western areas (Sun and Zhang, 2000). In the Taihu region, the watertable is high and all waterways (natural and man-made) are interconnected, which facilitates transport of excessive NO_3^- to the sea (Richter and Roelcke, 2000). However, the eutrophication of Taihu Lake accelerated from 1983/1984 to 1991. Beginning in 1983/1984, a rapid growth of township (small cities with populations of less than 3000) industries began. These rural industries, using outdated and inefficient technology, were considered to be the major contributors of untreated effluents (Sun and Zhang, 2000).

The NO_3^- level in Taihu Lake is well below the WHO-critical level for drinking water (50 mg NO_3^-/L, corresponding to 11.3 mg NO_3^--N/L (WHO, 1993)), but large scale non-point agricultural pollution is still contributing to the eutrophication and is thus threatening its use as a source of drinking water (Richter and Roelcke, 2000). However, in some regions in China, the NO_3^- content in groundwater and well water exceeds the critical WHO-level for drinking water (Zhu and Chen, 2002). In the 1980s, 38% of 76 investigated wells exceeded the critical value and in a recent survey on 40 samples, 27.5% had too high NO_3^- concentrations in Wu County, Jiangsu Province (Zhu and Chen, 2002).

Emissions of N to the atmosphere

The gaseous emissions that may have an agricultural origin, N_2O, NH_3 and NO_X, can have a serious environmental impact due to their involvement in the chemistry of the atmosphere. N_2O is an important greenhouse gas, which is also involved in the ozone equilibrium (Li and Lin, 2000). The global warming potential of N_2O expressed in CO_2-equivalents for a residence period of 100 years is 310 (CO_2 is the reference gas, assuming a global warming potential value of 1 unit). This value can be compared to the global warming potential of the greenhouse gas CH_4, which is 16-26 (See, 2001). Another greenhouse gas is NH_3, which is considered to be a contributor to the warming of the Earth. Emitted to the atmosphere, NH_3 interacts with OH radicals to produce NO_X (NO_2, NO) and in addition, it reacts with nitrate and sulphate to produce ammoniated aerosols. Other environmental problems connected with NH_3 emissions are soil acidification and groundwater pollution. As regards NO_X, the major damage is caused by the conversion of NO to NO_2, which is a primary source of tropospheric ozone (Li and Lin, 2000).

N_2O emissions from agricultural systems include direct emissions from agricultural fields (chemical fertiliser, animal manure, enhanced biological N-fixation through N-fixing crops, recycling of crop residues, sewage sludge application), emissions from animal production systems (animals themselves, wastes from confined animals, dung and urine deposited during the grazing period) and indirect emissions derived from N that originates from agricultural systems. The indirect emissions are: volatilisation and atmospheric deposition of NH_3 and NO_X; leaching and runoff; and human consumption of crops followed by municipal sewage treatment. These are considered to be the major pathways for chemical fertilisers and manure N inputs that give rise to indirect N_2O emissions (Mosier and Zhu, 2000). N emissions (N_2O, NO_x and NH_3) from agriculture are mainly caused by N fertiliser application, animal waste and biomass burning (Li and Lin, 2000). In a global perspective, Asia as a continent is a source of N to the rest of the world, as almost all of the anthropogenic N converted to N_2O and emitted to the atmosphere is transported out of Asia (Galloway, 2000).

The fractions of NH_3 volatilised and NO_X emitted from chemical fertilisers and animal wastes are influenced by soil, environment, fertiliser type and application technique. The NH_3 volatilisation from livestock manure is also affected by animal type and age, feed characteristics, C/N-ratio of the manure, climate conditions etc. Because of that, a lot of uncertainties are connected with estimations of the gaseous losses of these N-containing molecules (Li and Lin, 2000). Calculations of N emissions of agricultural origin in Asia in 1961 and 1994 showed that the relative importance of N_2O emissions from animal wastes, N-fixation and returning of crop residues to the field after harvest has decreased compared to the increased contribution from chemical fertilisers. The proportion from chemical fertilisers was 11 and 58% of the total N_2O emissions in 1961 and 1994, respectively. In addition, the direct NO_X emissions from the field have increased relative to the emissions from animal production (Mosier and Zhu, 2000).

Fertiliser application is suggested to be the most important source of N_2O from agricultural soils (Li and Lin, 2000). The emissions are enhanced by increased soil content of mineral N through increased nitrification and denitrification. In most soils, N application enhances the N_2O emissions (Mosier and Zhu, 2000). Of the total N input with chemical fertilisers, about 10-30 and 13-70% are emitted to the atmosphere after application to upland and paddy soils, respectively. Thus, the fertiliser use efficiency of the crop is low (Li and Lin, 2000). According to Zhu and Chen (2002) the gaseous losses of applied chemical fertiliser N are suggested to be about 45% from farmland, slightly higher than the gaseous losses in microplot field experiments (40% as median value). Under certain conditions, NH_3 may account for 40% of the applied N lost to the environment, and is then considered to be the dominant pathway of gaseous losses (Zhu and Chen, 2002).

According to Zhu and Chen (2002), the fertiliser use efficiency was halved from 1961-1965 to 1990-1993. It will decrease even more if advanced application technologies are not adapted in the future, when the limited arable land resource of China and the large population will increase the pressure on food production and probably cause an increase in fertiliser consumption (Li and Lin, 2000). Technologies that have shown emission reducing measures include, for example, point placement of urea instead of surface application and fertiliser incorporation to a sufficient depth (15 cm) (Richter and Roelcke, 2000). Historically, the increased N utilisation in Asia has increased the emissions of N_2O from agricultural systems by a factor of 2.6 during the period from 1961 to 1994 (from 0.8 to 2.1 Tg N_2O), with the most rapid increase in 1970-1990 (Mosier and Zhu, 2000). The direct emissions of N_2O from farmland have also increased during the period 1990-1995, from 282 Gg N in 1990 to 336 Gg in 1995. As regards the NH_3 volatilisation, losses from farmland have also increased during the same period, from 1.80 Tg N in 1990 to 2.71 Tg N in 1995. In 1995, the NH_3 loss from animal manure was 3.58 Tg N, giving a total loss of 6.29 Tg N (Zhu and Chen, 2002). However, the rate of increase in N input and emissions during the 1990s seem to have declined, but it could be questioned whether this is a long-term trend, or whether the increased food production pressure will accelerate the N input demand and result in increased emissions in the near future (Mosier and Zhu, 2000).

Management strategies may contribute to soil pollution

According to Wang and Tao (1998), wastewater from industrial and domestic sources introduces huge amounts of inorganic and organic contaminants into agricultural land through wastewater irrigation. Wastewater irrigation is considered as one of the most significant sources of soil pollution, and is a particularly acute problem in agricultural land areas adjoining large urban areas. In an investigation in the eastern outlying farming areas of Beijing, northern China, in which 75% of the site was paddy fields and the rest was used as vegetable gardens, the average of the eight heavy metals investigated were higher than those from the background agricultural soils, where wastewater irrigation was not commonly used. An analysis of the spatial distribution of the heavy metals showed that the concentrations were higher near irrigation outlets. Where sewage sludge had been periodically applied as fertiliser, the heavy metal content was even higher.

In Europe, the Cd (cadmium) content in chemical P fertilisers has become an urgent issue in later years, since Cd is considered to be a non-essential heavy metal that accumulates up in the food web and is toxic to human and animals. Some P fertilisers based on minerals mined in areas where sedimentation dominated the P containing rock formation are particularly high in Cd. When the fertiliser is applied to agricultural land, Cd is introduced as a contaminant. In Sweden, the main chemical fertiliser industry company, Hydro Agri, have introduced a 'Cd-guarantee' in their P fertilisers (homepage: Hydro Agri, 2004-01-13). The P-fertilisers produced in China are considered to have low amounts of Cd (ISSAS, 2003) and China produces more than 70% of the P fertilisers it consumes (Sheldrick et al., 2003).

Best management practice of nutrients to achieve a sustainable agriculture

In their review, Zhu and Chen (2002) proposed some best management strategies to increase food production and minimise the environmental impacts of N fertiliser use in China. Returning to the four general problems with the use of chemical N fertilisers mentioned in 'Fertiliser use in China', above, namely: too high rates of N application, improper application methods, N fertilisers applied too early and an unbalanced

N:P:K-ratio, the following were thought to achieve a more sustainable agriculture from a nutrient point of view:

Optimising the N application rate

The maximum crop yield is usually achieved by higher N application rates than is economically most profitable considering the cost of fertiliser versus the increase in yield per kg fertiliser applied. Thus, a lower fertiliser input than currently would be more profitable from the farmer's point of view. However, the N application rates used are considerably higher than what is economically justified. This approach leads not only to lower income, but also to increased losses of nutrients, since the N use efficiency is decreased by increased N input. By using recommendations on the mean application rate of N, which is a rough estimate of the need for fertiliser N, the extension technicians will have a reference in their advisory work. One merit with the method is that it can be easily adopted in rural areas where soil-testing facilities are not available and the cropping index is high. Of course, the recommendation must be adjusted according to the local conditions, such as irrigation and variety of crop. The accumulation of soil mineral N after harvest can be used as an indicator of a too high N application rate (Zhu and Chen, 2002).

Deep placement of fertilisers

In micro-plot experiments, the plant recovery of chemical fertiliser N was increased by 23-37% by deep placement of supergranular urea instead of surface-broadcast or followed by incorporation. Along with the increased recovery, a substantial N loss reduction, from 30-53% to 11-21%, was observed. The same effect was experienced in field investigations (Zhu and Chen, 2002).

Matching the N application time to crop demands

Field micro-plot experiments have revealed that the N recovery in rice plants was dependent on the growth stage at which the N fertiliser was applied. In the early growth stage, the N recovery of applied fertilisers was 22-52%. It increased to 55-69% at vigorous growth stages and thus the N losses were reduced. By splitting the fertiliser application in rate and time, the application rate of fertiliser N could be reduced considerably without any loss in yield (Zhu and Chen, 2002).

Balanced fertiliser application

By a balanced fertiliser use, the agronomic N use efficiency can be improved and the accumulation of mineral N in soils and the N losses can be reduced (Zhu and Chen, 2002). In this context, one must remember the importance of balance between N, P and K input, as well as that of other nutrients (Jin et al., 1999). The P balance is, according to Zhu and Chen (2002), "remarkably positive" (no figures given), while the net balance in K fertilisation is negative. However, Sheldrick et al. (2003) found that there was a net depletion of P at the national level in 1997, corresponding to -0.6 million tons P/year. The K-depletion rate (calculations include K in manure) at the national level was 62 kg/ha and year in 1991 (Sheldrick et al., 2003). In some regions, pronounced effects of K fertilisers have been observed. Some paddy soils in the northeast region of China are deficient in S, and bioassays with soil samples from 17

Chinese provinces indicate that deficiencies in S, Ca, Mg, Zn, B, Mo, Cu and Mn are fairly widespread (Jin et al., 1999).

Nitrification inhibitors or controlled N release fertilisers

By using nitrification inhibitors, the denitrification losses of applied N fertiliser can be reduced by 21-27% and by inhibiting the nitrification of ammonium, the leaching losses of NO_3^- and the emissions of N_2O can be assumed to decrease. The use of nitrification inhibitors and controlled released fertilisers, for example urea coated with a thin organic film, does not require any extra labour for implementation, and may thus be attractive to farmers. A limiting factor for a wider application is the low cost of existing N fertilisers, which challenges the attractiveness of the excessive additional costs of the 'improved' fertilisers (Zhu and Chen, 2002).

Zhu and Chen (2002) also set up seven recommendations to achieve an increased food production and minimise the environmental impacts:

1. Increase food production in the low- and mid-yielding areas to meet the population growth.
2. Balance the yield target of crop production and environmental consequences in high yielding areas.
3. Improve farmland infrastructure (irrigation, drainage, terracing), eliminate the limiting factors in soils (nutrient deficiency, water logging), and increase soil productivity in low- and mid-yielding regions.
4. Balance the NPK ratio by increasing the rate of K fertilisation.
5. Strengthen the extension service with strong environmental consciousness and promote more efficient N management with assistance of computer modelling.
6. Encourage the use of organic manure in agriculture and reinforce treatment of civil sewage water and wastewater of intensive livestock farm.
7. Future research on N management based on the principle 'High quality, high efficiency, high yielding, low cost and minimum impact on the environment'.

GLEAMS-simulations of N and P losses from an intensive vegetable-farming system in Shiba village, a peri-urban area of Nanjing, Jiangsu province, China

Background

During recent decades, a rapid urbanisation has occurred in China. Because of improper handling of urban wastes, the rapidly expanding areas now face environmental problems (Duan et al., 2000; Sun and Zhang, 2000). Wastewater from industrial and domestic sources is considered to be one of the most significant sources of soil pollutants (Wang and Tao, 1998). Heavy loads of this polluted water and sewage directly into rivers cause severe problems in rivers and lakes along the coast (Duan et al., 2000; Sun and Zhang, 2000). When the contaminated water is used for irrigation, large amounts of inorganic and organic pollutants, including heavy metals, are introduced into agricultural land. The problem is particularly acute in areas used for agricultural purposes adjoining large urban areas (Wang and Tao, 1998). China and Vietnam, which also suffers from similar problems, are now at an early stage of identifying the different pathways for and impact of wastes into the peri-urban areas, but appropriate directives and risk assessments are still missing (RURBIFARM, 2002).

In peri-urban areas, rice-based farming systems have been replaced with intensive vegetable production, regarded as highly profitable due to the closeness to the markets. However, the vegetable-producing systems, which enable 2-3 harvests a year, require large amounts of inputs (nutrients and pesticides) to keep the yields up, resulting in risks for significant losses of nutrients from the system as fertilisation exceeds the nutrient uptake of the harvested crops (RURBIFARM, 2002).

According to Jin and Jang (2002), the key to a sustainable development of crop production in China, where a large, ever increasing population is putting pressure on its limited agricultural land resource due to an increased demand for food, is management of soil nutrients and an efficient allocation of inorganic and organic resources of nutrients. The RUBRIFARM-project; 'Sustainable Farming at the Rural-Urban Interface – An Integrated Knowledge Based Approach for the Nutrient and Water Recycling in Small-scale Farming Systems in Peri-urban Areas of China and Vietnam', aims to contribute to the development of more sustainable interaction in the rural-urban interface through effective recycling of by-products, wastes and water (called 'urban wastes') in peri-urban farming systems, based on sound risk assessments and policies related to human health and environmental impact. This is proposed to be achieved by developing an integrated knowledge based system (IKBS), combining the goals of food safety, low environmental impact and high profitability of farmers (RURBIFARM, 2002).

The RURBIFARM-project is based on 5 work packages (WP), where WP 1-4 are set up to deliver results and methods for both strategic and applied research, while WP 5 is more extension related. In WP 4, called 'Reducing environmental risks – soil and water contamination. Risk assessment', the main objectives include modelling states and fluxes of water, nutrients, pesticides and heavy metals at field level in current farming systems and evaluating the environmental impact of current farming systems concerning the use of urban wastes, pesticides and fertilisers in short- and long-term perspectives and on field and catchment scale. The researchers will also develop a protocol for threshold values to be used for risk assessments of environmental impact on soil and water quality and model and evaluate scenarios of novel farming systems in time and scale. The use of simulation models enables extrapolation of the changes in time and space. N, P and pesticide fluxes will be in focus for the simulations, due to their strong eutrophication and contamination effect on surface waters and groundwater (RURBIFARM, 2002).

The process-based simulation model GLEAMS (Groundwater Loading Effects of Agricultural Management Systems, Knisel et al., 1999), which simulates N, P and pesticide fluxes on field level, will be used to estimate factors governing the utilisation efficiency of added nutrients by the vegetables grown versus losses to the environment. In combination with other tools, the environmental impact at catchment scale will be estimated (RURBIFARM, 2002).

In this report, I describe use of the GLEAMS model to simulate the N and P fluxes in a vegetable-producing peri-urban area near Nanjing, China. The outcome of different practices of nutrient management was studied to identify the events causing the major environmental impacts, as well as the most effective ways of nutrient management.

Site description

One of the peri-urban areas studied in the RURBIFARM-project is Maqun Township, 5 km east of Nanjing City in Jiangsu Province, China (ISSAS, 2003), situated in the Northern subtropical monsoon area. In this area, there are four distinct climatic seasons, with an annual mean temperature of 15.7°C and an annual frost-free period of 237 days (K. Blombäck, pers. comm). The mean annual rainfall in this area is 1100 mm, concentrated to between May and July (ISSAS, 2003) and the annual mean evaporation is 1563 mm (K. Blombäck, pers. comm.). The soils are classified as Hortic Anthrosols, consisting of clay loam to clay (ISSAS, 2003).

Maqun Town consists of nine villages and has 23 580 inhabitants of which 7 837 are farmers. The area of arable land is about 200 ha, and most of it is used for vegetable production. The village chosen for the farming system survey, Shiba Village, covers nine groups, 400-440 households with a total population of about 1200, 80% of which get their income from vegetable production. Of the 55 ha of arable land in the village, a total of 35 ha is used for vegetable production. Until 1964, rice was the predominant crop in the area. Between 1964 and 1972 the farmers started to plant vegetables on half of the area, while rice was still planted on the other half. The farmers then all changed to vegetable production. Until 1985, all inputs (seeds, fertilisers, pesticides) and the market were managed by the village committee, but since 1985, the farmers have to buy inputs as well as sell their production themselves at the local market. Since 2000, 5-6 immigrant households from Sichuan and northern Jiangsu Province have rented land from local farmers to engage in vegetable production (ISSAS, 2003).

Each household in Shiba Village consists of 3-8 members, among which 1-3 members are working in the city or in town. Generally, the farm labour in each household is 40-60 years old, except in the immigrant households, where they are generally younger. For the immigrants, the only income source is the vegetable production and a larger area of arable land per household is managed by them than by the local farmers (ISSAS, 2003).

Each local household has approximately 2-3 mu (1 mu = 1/15 ha), corresponding to 0.13-0.53 ha of arable land, with 3-6 plots distributed at different places. The area for vegetable production depends on the distance away from the city, and increases with an increase in distance. Two to three, in some cases even 5 successive vegetable crops per year are planted in the area, the numbers of crops increasing with the distance away from the city. Some common vegetable rotations in the area are pepper-celery-lettuce, bitter melon-lettuce or kidney bean-cauliflower-spinach. The planted vegetable types include leaf vegetables (bok-choy, cabbage, celery, lettuce, spinach), fruit vegetables (tomato, bitter melon, egg-plant, pepper), beans (long bean, kidney bean, soybean and pea), root vegetables (carrot, radish, potato, sweet potato) and others (cauliflower, broccoli etc). Pesticides are applied regularly, and the fields are irrigated with water from ponds near the plots or from the river. For the whole Maqun Town, the annual total yield of various vegetables is reported to be 0.27-0.3 million kg. The income from the vegetable production in Shiba village is 10000 yuan/household and year, or 4600 yuan/farmer and year, somewhat higher than the mean income of 4000 yuan/vegetable farmer and year for Maqun Town (ISSAS, 2003).

For fertilisation, up to 2 tons fw (fresh weight) cow manure/mu and year (30 t (fw)/ha and year) from a dairy farm in the vicinity of the city has been applied each spring since 1985. Also, 100 kg/mu and year (1500 kg/ha and year) of composite fertiliser (NPK 10-6-9) is used as basic fertiliser, applied before the first vegetables of the year are planted. During the season, 80-100 kg/mu (1200-1500 kg/ha) of urea (46% N) together with human manure are used as instant fertiliser, in 4-6 applications for each of the vegetable seasons (ISSAS, 2003).

The farmers sell their products as wholesale or retail, preferably as retail in the market in the town and in the city since the price is higher than wholesale. However, the farmers have to pay rent for a counter (80 and 120-150 yuan/month in town and city, respectively) for the retail. For the wholesale, the products are sold to dealers in the field. The proportion of wholesale to retail increases with area planted (ISSAS, 2003).

A survey of the environmental situation shows no great risk of heavy metal pollution in surface water in the peri-urban areas of Nanjing (ISSAS, 2003). However, the total N and P concentrations in surface waters were high. High levels of P in the waters may come from rivers receiving wastewaters from the city and runoff from heavily fertilised fields. It was also found that Cd and Zn contents were higher in soils in plots near the roads, where larger amounts of cow manure are applied, compared to plots farther away. Analyses of the native P fertilisers in the 1980s showed that the heavy metal content was low, in contrast to some of the imported P fertilisers. Due to P shortage in China, however, a lot of P-fertilisers are now imported, resulting in increased attention being paid to the potential risk of heavy metal contamination of soils from the chemical fertilisers (ISSAS, 2003).

Methods

To estimate the N and P losses in an intensive vegetable production system in Shiba Village, the GLEAMS model was used. To achieve a basic parameterisation of the model, general data available from various sources were collected and adapted for model use, with only few site-specific data on management practices and field and soil properties (ISSAS, 2002, 2003; Blombäck and Djodjic, 2003). The basic parameterisation formed a possible scenario for the Shiba site, using Nanjing climatic data, for a pepper-cabbage-lettuce crop rotation (three vegetable crops/year) during an 11-year period. The simulated results were used to study the current situation with respect to nutrient use efficiency and losses of N and P and were also compared with simulated scenarios where the nutrient management or crop rotation was alternated. For all simulated scenarios the respective N and P use efficiency were calculated. Since K is not included in the GLEAMS model, it was not simulated, but the choice of fertiliser will have an influence of the K balance in the cropping system. The formula used was:

$$\text{Nutrient use efficiency} = \frac{\text{nutrient yield in harvest}}{\text{fertilisation} + \text{mineralisation}}$$

Some of the on-farm activities causing environmental impacts could then be identified, and strategies to increase the nutrient use efficiency by changed management and to control N and P losses from the field could be proposed. Data on crop rotation and management were provided by Dr. Karin Blombäck (pers. comm) and ISSAS (2003).

Scenario 1: Current crop rotation and management

The crop rotation, planting and harvesting dates, tillage activities and manure/fertilisation applications used by the GLEAMS model during one year in the simulated 11-year period are shown in Figure 3 and in Table 4. All the years in the simulated period were similar in terms of crop rotation and management strategies. Pesticide use was not considered and not simulated, either in this scenario or the following.

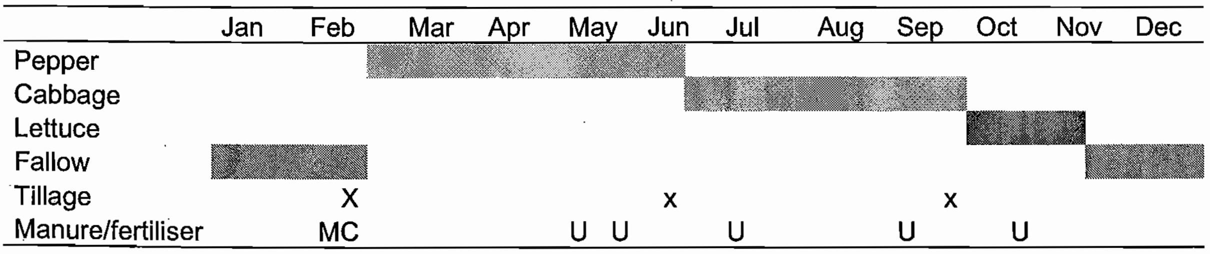

Figure 3. Crop rotation and management strategies for one field in Shiba Village, Maqun Town, Nanjing during one year. x= tillage operation, M= manure application, C= application of composite fertiliser (NPK 10-6-9), U= application of urea.

Table 4. The approximate Julian day* of the year of planting, tillage, manure and chemical fertiliser application (NPK, urea) and harvest in scenario 1, the current crop rotation and management in Shiba Village**. Tillage and incorporation depth of manure and NPK was 15 cm. Tillage operations were performed using a row cultivator.

	Planting	Manure application	NPK-application	Tillage	N-fertiliser (urea)	N-fertiliser (urea)	Harvest
Pepper	40	41	41	42	135	180	195
Cabbage	196			197	200	255	270
Lettuce	271			272	300		335

*Julian day: the number of days after January 1 (e.g. day 40 is Feb 9).
**Due to limitations in the model, the tillage and the manure and fertiliser applications was done after planting of the crop, while in the reality, these activities are done before planting.

The basic fertilisers, 30 tons (fw)/ha and year of beef manure (not authentic to Shiba Village, where cow manure from a dairy farm in the vicinity is used) and 1500 kg/ha and year of the chemical composite fertiliser NPK 10-6-9 were applied in the end of the fallow period, just before the tillage operation. The tillage involved hand digging, resulting in about 15 cm tillage depth. Since no hand digging option is included in the GLEAMS model, the hand digging operation was assumed to be similar to tillage using a row cultivator. After the tillage operation, pepper seedlings were planted. Urea (46% N) and human manure were used as instant fertilisers, but as the amount of human manure spread on the field was small, it was not taken into account in the simulations. Each application of urea consisted of 630 kg NH_4-N/ha, corresponding to a urea rate of 1370 kg/ha. Urea was applied twice during the pepper season. After harvesting of the pepper, the soil was tilled (15 cm deep) and cabbage seedlings planted. As for pepper, urea was applied twice. When the cabbage was harvested the soil was tilled again (15 cm deep) and lettuce seedlings were planted. Urea was applied once. After the lettuce was harvested, the soil was left fallow until the next vegetable crop.

Scenario 2: 'Best management practice' (BMP) – reduced amount of fertilisers

In the second scenario, the amounts of organic and chemical fertilisers were reduced compared to the current situation described above. Crop rotation and tillage operations were in accordance with the present management strategies as shown in Figure 3 and Table 4, above.

Assuming an annual N removal of 300 kg N/year in the harvested crops (based on a total N removal of 3265 kg N for the simulated 11-year period in scenario 1) and an annual P removal of 40 kg/ha, the new amounts of fertilisers was determined by the replacement method of nutrient application. This was based on the amounts of nutrient removed with the harvest. In order to secure the yields, the simulated application rate for this scenario was twice the annual N and P yields in the harvest, giving an application rate of 600 kg N and 80 kg P/ha and year.

Assuming that the P content in the manure was 1.6% on a dry weight basis, and that twice the P demand of the vegetables would be applied with the manure, calculations showed that 5.1 tons dry manure/ha and year, corresponding to 25.5 tons of manure (20% dry matter)/ha were needed. The P supply with the amount of manure needed was thus 82 kg/ha. Consequently, no composite fertiliser was necessary to meet the crop P demand, and was thus not applied in the second scenario.

The GLEAMS model assumes that the N-content in beef solid manure (15-25% dry matter) is 4.8% on a dry weight basis. With the 25.5 tons manure/ha spread to cover the P demand of the crop, 245 kg N/ha was applied. Thus, the need for 'extra N fertilisation' with the instant fertiliser was 355 kg N/ha. In this scenario, urea was applied on five occasions at the same times as in scenario 1. Each time, 71 kg NH_4-N/ha, corresponding to 154 kg urea/ha, was spread.

A summary of the annual nutrient input to each of scenarios 1-4b is shown in Table 5.

Scenario 3: Change of instant fertiliser

The total N content in ammonium nitrate (NH_4NO_3) is 33%, with equal amounts of NH_4-N and NO_3-N. Urea, 46% N in total, consists only of NH_4-N. According to the Swedish Agricultural Board (SJV, 2003), for agricultural crops such as cereals, ley, rape, sugarbeet and potatoes, N losses to the environment by leakage and denitrification during periods of excessive precipitation can be reduced if the NH_4-N content in the fertiliser is high. However, NO_3-N dominated fertilisers are preferred in order to increase the fertiliser use efficiency (SJV, 2003; L. Mattsson, pers.comm.) of instant fertilisers in 'growing crop', since NO_3-N is more easily absorbed by the plants. In Sweden, the commercial fertiliser recommended for the latter purpose is calcium nitrate ($Ca(NO_3)_2$, (16% N) (SJV, 2003), but in the international perspective, this fertiliser is not commonly used. The hypothesis for the third scenario was that if the amount of N supplied by urea is replaced with the same amount of N from ammonium nitrate as an instant fertiliser, the N efficiency will increase and thus, the N losses from the field will decrease. Thus, a total of 355 kg N/ha was applied each year, corresponding to 1075 kg NH_4NO_3/ha, divided equally among 5 occasions (71 kg N/occasion of which 35.5 kg was in the form of NH_4-N and 35.5 kg N in the form of NO_3-N). All other variables, such as crop rotation, total N and P amounts, number and date of fertiliser applications and tillage operations were similar to the 'best management practice' in scenario 2, described above.

Scenario 4a: Changing the crop rotation - Turnips

In the fourth scenario, the crop rotation was assumed to be changed by replacing cabbage with radish (*Raphanus sativus* var. *longipinnatus*), as radish is also commonly planted in Shiba Village during the same vegetable season as cabbage (ISSAS, 2003). The hypothesis for this simulation was that a vegetable crop with a deeper root system would be more effective in absorbing excessive nutrients from deeper soil layers than a crop with a shallow root system. According to Runåbergs fröer (1999), a Swedish seed company, radish has a taproot that is 25-50 cm long and 4-7 cm in diameter. Radish is reported to be easy to grow and the recommended density of plants is 15-20 cm between each radish planted in rows with 35-50 cm between rows (Runåbergs fröer, 1999). However, the crop characteristics of radish were not included in the GLEAMS model database, and no data on either the potential yield or the nutrient content in the vegetable were available. Thus, the characteristics of radish had to be estimated. Assuming that apart from root depth, all crop characteristics such as potential yield, dry matter content, ratio of total N to P and C/N ratio at harvest were identical for radish and turnips, turnips were used for the simulations. Turnips is also a brassica (Compositae), but with a shallower rooting system than radish (Runåbergs fröer, 1999). A more preferable scenario would have been to simulate radish using the same plant characteristics as for turnips but with increased rooting depth. However, I was not successful in adjusting the rooting depth in the model.

With turnips instead of cabbage, 'pilot simulations' showed that the total N yield was reduced to 2830 kg N/ha in total, or 257 kg N/ha and year and the total P yield to 309 kg/ha, corresponding to 28 kg P/ha and year. Thus, the fertiliser input had to be decreased. Even in this scenario, the replacement method was used as a basis for the fertilisation rates. Double the crop N and P demand was assumed to be an adequate supply to secure the yields.

With 4.8% N and 1.6% P in the solid beef manure (dry weight basis), calculations showed that 17.5 tons of solid manure consisting of 20% dry matter would supply twice the P demand and 168 kg of the necessary 514 kg N/ha. Thus, 346 kg N/ha was applied as instant fertiliser (urea) on 5 occasions (69 kg N/ha and occasion). Except for the crop rotation and fertiliser input, all other variables in management strategies were in accordance with scenario 2, 'best management practice', above.

Table 5. Annual nutrient input to an intensive vegetable cropping system in Shiba Village for scenarios 1-4b, simulated for an 11-year period. For manure and composite fertiliser, the total amount applied as well as the total amounts of N and P it contained are shown. For urea and NH_4NO_3, only total amount of N is included. All values except manure are in kg/ha and year. Application rate of manure (fresh weight) is given in tons/ha

Input	Scen. 1[1]	Scen. 2[2]	Scen. 3[3]	Scen. 4a[4]	Scen. 4b[5]	
Manure, fw	30.0	25.5	25.5	17.5	19.4	tons/ha and year
N	288	245	245	168	186	kg N/ha and year
P	96	82	82	56	62	kg P/ha and year
Urea	5x630	5x71		5x69	5x58	kg N/ha and year
NH_4NO_3			5x71			kg N/ha and year
NPK 10-6-9	1500					kg/ha and year
N	150					kg N/ha and year
P	135					kg P/ha and year

[1]Scen. 1: The current situation
[2]Scen. 2: Best management practice - reduced nutrient input
[3]Scen. 3: Best management practice - changed instant fertiliser
[4]Scen. 4a: Best management practice - changed crop rotation (turnips)
[5]Scen. 4b: Best management practice - changed crop rotation (eggplant)

Scenario 4b: Changing the crop rotation - Eggplant

In this scenario, cabbage was replaced by eggplant in the crop rotation. Eggplant is also a common crop in the Shiba Village and is included in the GLEAMS model crop characteristics database. Eggplant has a deeper root system than cabbage, which is accounted for in the model. The hypothesis was the same as in scenario 4a, and fertilisation rate was chosen in accordance with twice the replacement of harvested nutrients theory, as in scenario 2-4a. Pilot simulations showed that the total removal rates of N and P in harvested crops were 2622 kg N and 339 P kg/ha, respectively, for the simulated 11-year period. Thus, the annual removal rates were 238 kg N and 31 kg P/ha, corresponding to a fertilisation rate of 476 kg N and 62 kg P/ha. Calculations showed that 19.4 tons of solid manure (20% dry matter) would be needed to cover twice the crop P demand. The manure would also introduce 186 kg N/ha to the system. Thus, 290 kg N/ha was annually needed as instant fertiliser, corresponding to a fertilisation rate of 58 kg N/ha on each of 5 occasions, applied as urea. As in scenario 4a, all other variables on management strategies were the same as in scenario 2.

Results and discussion

The annual nutrient input, uptake, losses and transformations of N and P as averages of each of the scenarios 1-4b for the 11-year simulation period are summarised in Tables 6 (N), and 7 (P). In Figure 4, the annual N losses and transformations are shown. The manure applied is included in the mineralisation term.

Table 6. Annual N input, output, losses and transformations (kg N/ha and year) as averages for an 11-year simulation period of an intensive vegetable production system in Shiba Village. No irrigation was used. Input terms: fertilisation and mineralisation. Output: N in harvested crop. Crop uptake is considered as a transformation, since the N that is not taken away with the harvest (output) is mineralised (input) later. Losses are denitrification, runoff, leaching and ammonia volatilised. The N not leached as NO_3^- was leached as NH_4^+ (small amounts)

	Scen.1[1]	Scen. 2[2]	Scen. 3[3]	Scen. 4a[4]	Scen. 4b[5]
Fertilisation	3300	355	355	345	290
Mineralisation	1076	954	953	633	803
Crop uptake	480	480	480	368	448
N yield in harvest	297	297	297	257	238
Denitrification	2036	471	486	337	353
Runoff	390	90	74	70	74
Leached, total	663	106	108	102	80
Nitrate	663	106	108	102	79
Ammonia volatilised	45	38	38	26	29
N use efficiency[6]	0.07	0.23	0.23	0.26	0.22

[1]Scen. 1: The current situation
[2]Scen. 2: Best management practice - reduced nutrient input
[3]Scen. 3: Best management practice - changed instant fertiliser
[4]Scen. 4a: Best management practice - changed crop rotation (turnips)
[5]Scen. 4b: Best management practice - changed crop rotation (eggplant)
[6] N use efficiency = N yield in harvest/(fertilisation+mineralisation).

Table 7. Annual P input (fertilisation, mineralisation), output (P yield in harvest), losses (runoff, leaching) and transformations (crop uptake) (kg/ha and year) as averages for an 11-year simulation period of an intensive vegetable production system in Shiba Village

	Scen. 1	Scen. 2	Scen. 3	Scen. 4a	Scen. 4b
Fertilisation	135.00	0.00	0.00	0.00	0.00
Mineralisation	558.83	405.09	405.07	261.77	326.62
Crop uptake	48.43	48.50	48.50	38.87	58.18
P yield in harvest	30.34	30.39	30.39	28.07	30.81
Runoff	9.11	4.86	4.86	3.55	5.35
Leached, total	0.08	0.08	0.08	0.08	0.08
P use efficiency[6]	0.044	0.075	0.075	0.107	0.094

[1]Scen. 1: The current situation
[2]Scen. 2: Best management practice - reduced nutrient input
[3]Scen. 3: Best management practice - changed instant fertiliser
[4]Scen. 4a: Best management practice - changed crop rotation (turnips)
[5]Scen. 4b: Best management practice - changed crop rotation (eggplant)
[6]P use efficiency = P yield in harvest/(fertilisation+mineralisation).

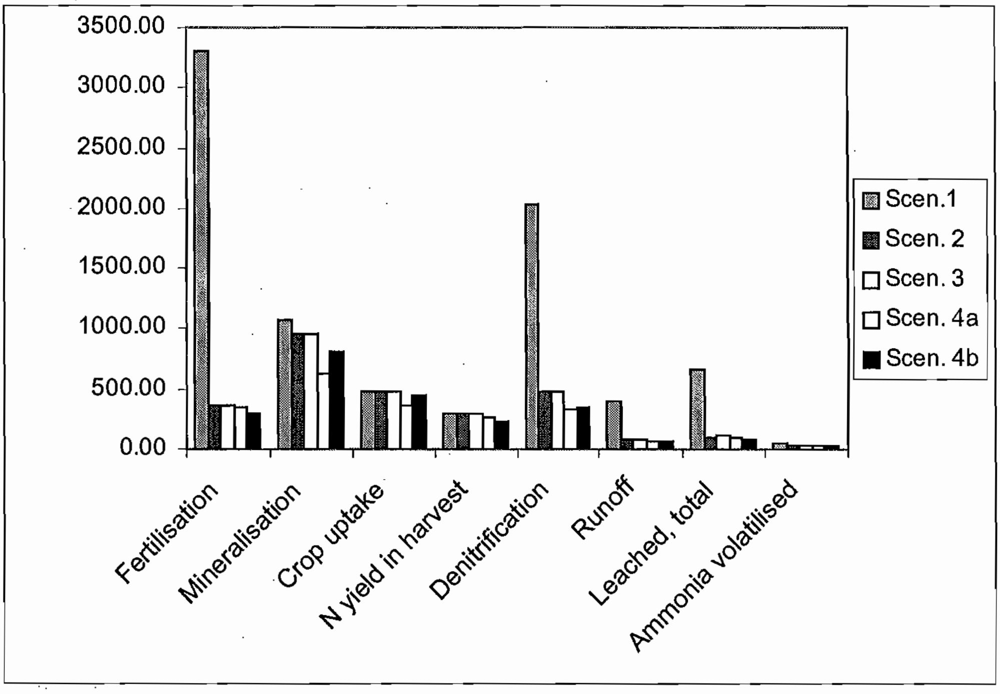

Figure 4. Annual N input, output, losses and transformations (kg N/ha and year) as averages for an 11-year simulation period of an intensive vegetable production system in Shiba Village. No irrigation was used. Total losses by leaching consist mainly of NO_3^-. Scen. 1: The current situation, Scen. 2: Best management practice – reduced nutrient input, Scen. 3: Best management practice – changed instant fertiliser, Scen. 4a: Best management practice - changed crop rotation (turnips), Scen. 4b: Best management practice - changed crop rotation (eggplant).

After the simulation of scenario 1, it was obvious that something must be done to reduce the nutrient input to the system since there was an enormous surplus of nutrients that were lost to the environment. The most effective way of decreasing nutrient losses in this intensive vegetable-cropping system is probably to reduce the fertiliser input. Thus, the later simulations (scenarios 2-4b) were based on the fertilisation rate of twice the nutrients in the harvest principle. In scenario 3-4b, I tried to improve the nutrient use efficiency and reduce the nutrient losses to the environment by small changes in other management factors, with the second scenario as a base.

One might think that reducing the losses to zero would be the best approach to achieve a sustainable system, but it is not that simple. In some cases, fertilisers are needed not only to secure a high yield, but also to achieve a good crop quality. When the products are to be used for special purposes, commonly in the food industry sector, there are high demands on, for example, the protein content in wheat for baking and barley for malting, N and K content in sugarbeet and Cd levels in baby food, which must be fulfilled (Fogelfors, 2001). In these cases, a proper nutrient balance in the fertiliser input is a must. Therefore, a balanced nutrient management that secures long-term sustainability as regards soil fertility, high yields, high quality, socio-economic aspects and minimisation of environmental impact is more desirable than a nutrient management where these aspects are overlooked in order to reduce the nutrient losses to nil.

Reducing the fertiliser input to the fields in Shiba Village may not only result in decreased environmental impact due to decreased N- losses, but may also contribute to reduced attacks by some pathogens and insects and thus the pesticide input, which is also very high, can be lowered. Increased use and amounts of fertilisers, especially N, are generally considered to increase the severity of pathogens that prefer young, succulent tissues while the diseases caused by pathogens that attack primarily mature or senescent tissues are decreased (Agrios, 1997).

When I began to summarise and interpret the results from the simulations, I was confused when I found that the nutrients supplied by manure were not included in the 'fertilisation' input term. The GLEAMS model includes the nutrients in organic fertilisers in the 'mineralisation' term, despite the fact that some of the nutrients in organic fertilisers are readily available to the crops. Thus, when calculating the nutrient use efficiency, I decided to calculate it as 'nutrient yield in harvest/(fertilisation+mineralisation)' instead of 'nutrient uptake by the crop/(fertilisation+mineralisation)' to avoid double-counting of the crop residues mineralised (both as input and output terms) or as 'nutrient yield in harvest/fertilisation', since this approach would exclude the nutrients supplied with the manure.

Scenario 1: Current situation

Assuming that the simulations of the first scenario are not very far from the truth, the current situation in Shiba Village is quite frightening from a nutrient management – sustainable agriculture point of view. Both N and P use efficiencies, as I calculated them, are very low, 0.07 and 0.04 for N and P, respectively. The N fertiliser input rate is very high, 3300 kg N/ha and year, which is far from the 300 kg N/ha harvested in the vegetables. According to J. Linder (pers. comm.), an annual surplus of 0-200 kg N/ha could be considered as normal in Swedish agriculture, with the highest values for farms with high animal density. In the simulated scenario, most of the applied N is lost to the surroundings by denitrification, leaching and runoff, probably with severe consequences for the environment.

The situation is slightly better for P, but there is still a loss, resulting from runoff of 100 kg/ha for the simulated period, corresponding to an annual P loss rate of 9.1 kg/ha. Liu et al. (2003) found that the biological activity was strongly P inhibited in the Changjiang River, in whose catchment Shiba Village is included. Decreasing the P losses is a main issue to avoid eutrophication problems in the main river of Changjiang, its tributaries and its estuary. Since the leaching rate of N is very high, a slight increase in P concentration in the water may cause a massive increase in the biological activity, thus enhancing the eutrophication rate.

When interpreting the results of the simulations, one must also consider the crop yields in the model. As all crop characteristics, including the potential yields, are supplied from the GLEAMS model database, it is possible that the yields in Shiba Village are either higher or lower than in the model. Based on the information that the vegetable production systems are very intensive in the area, not only with regard to the nutrient and pesticide inputs and number of crops per year, but also the space between plants and rows is much smaller than what is common in for example, Europe and North America, the yields may thus be higher than simulated. In that case, higher rates of fertiliser input may be justified and the nutrient losses may be lower than those simulated in scenario 1.

Another approach that must be taken into account is the probability that the same crop rotation would be used at the same field for eleven years without any interruption. From an agronomist's perspective, that is not recommendable, particularly because it facilitates the spreading of diseases and other pests. However, the annual averages of the nutrient transformations and losses are more secure if the simulated period is long than if it is only one or two years.

Scenario 2: Best management practice

In the second scenario, the fertiliser input was significantly reduced, in the case of N from an annual average of 3300 kg N/ha to 355 kg N/ha, without any decrease in crop yield. This improved the nutrient use efficiency and decreased the nutrient losses from the system (the field), especially for N, which was exactly as expected. The N use efficiency increased to 0.23, 3.3 times compared to that in scenario 1, and both the denitrification and N losses by leaching decreased, by factors of 4.3 and 6.2, to 471 and 106 kg N/ha and year, respectively. The decrease in ammonia volatilisation may be explained by the reduced manure application rate, which was 25.5 tons/ha (fw) and year in this scenario.

The P losses and transformations were also reduced when the P fertiliser input was decreased. However, as discussed above, the input term 'fertilisation' in Tables 6-7 above does not include the P supplied by organic fertilisers, here beef solid manure. The P use efficiency was improved in scenario 2, as was the nutrient balance. The P runoff was decreased to 4.9 kg/ha.

Whether an exclusion of a composite chemical fertiliser, as in scenario 2, is recommendable in a long-term sustainability perspective can be discussed. As described by Sheldrick et al. (2003), most soils in China suffer from increasing K depletion rates, since K input from fertilisers and recycled crop residues are relatively small compared to the output in harvest (see literature review section). The authors' conclusion was that China's future food security is seriously threatened by the high soil K depletion rate found, and that an increase in K fertiliser use as well as improved management of crop residues would be necessary to reverse the trend and achieve K balance. With high rates of K depletion, such as in Jiangsu Province, even soils with high levels of exchangeable K were expected to be serious affected in a medium time perspective (Sheldrick et al., 2003).

According to SJV report 2003:22 (2003), the K effect of 10 tons/ha evenly spread solid manure or slurry (10-20% dry matter) and urine (1-5% dry matter) from dairy cattle is 40 and 50 kg K/ha, respectively. In scenario 2, the annual application rate of solid manure was 5.1 tons/ha (dry weight), corresponding to an amount of 102 kg K/ha, assuming that the dry matter content in the manure is 20%. The reduction of K input by not using any composite fertiliser is 135 kg/ha and year, assuming that the amount of K input with the manure is the same as in scenario 2 and that the application rate of composite, NPK 10-6-9, is 1500 kg/ha and year. The amount of K in the harvest varies with crop (species and cultivar) and yield. For Swedish agricultural dicotyledonous crops, amounts from 16 kg K/ha for rapeseed (harvested yield 2 tons/ha) to 150 kg K/ha for potatoes (yielding 30 tons/ha) and sugarbeet leaves (30 tons/ha) have been reported (SJV, 2003). One must remember that in Sweden, the season when cultivation is possible is short, and only one crop per year is grown so greater amounts of K will probably be taken from the field with the harvested crops in the intensive multi-cropping system of China. If that is true, the decreased K fertilisation will severely threat the soil fertility in the future.

Another aspect of K fertilisation that must be considered when cultivating vegetables for sale is the quality of the products. According to Havlin et al. (1999), crops deficient in K show visual symptoms such as white spots (alfalfa) or chlorosis and necrosis on leaf edges (grasses), straw weakening of grain crops and increased degree of crop damage by pests. Plants suffering from K deficiency may also show wilting symptoms and reduced growth, since K is important in the ATP-production and transport of sugar and is involved in the opening and closure of stomata. The visual symptoms of K deficiency probably decrease the attractiveness of vegetable products to consumers at the market. It is also important to remember that monocotyledonous crops such as cereals are more efficient in absorbing K from the soil compared to dicotyledonous crops such as vegetables (Havlin et al., 1999). Thus, dicotyledonous crops have a higher demand on high soil K levels and symptoms of K deficiency may occur at an earlier stage.

Scenario 3: Change of instant fertiliser

Replacement of the NH_4-N dominated urea by a fertiliser in which half the N-content consisted of NO_3-N instead was thought to further reduce the N losses. NO_3-N is readily available and easier to absorb by the crop directly after fertiliser application compared to NH_4-N. Thus, the N losses may decrease since less N would remain in the soil after the last vegetable season. However, NO_3^- is more susceptible to leaching than NH_4^+ due to a higher solubility.

The hypothesis of improved nutrient use efficiency by changing fertiliser type was shown not to be true according to the GLEAMS simulation results in scenario 3. The N yield in harvest as well as the crop uptake were the same as in scenario 2, which may be an effect of the equations the GLEAMS model uses for calculations. The GLEAMS model does not make any distinction between different forms of mineral N when considering the crop uptake rate. The main pathway of N loss was still denitrification, which increased by the change of fertiliser type: from 471 kg N/ha and year in scenario 2 to 486 kg N/ha and year in scenario 3. The N lost by leakage increased slightly to 108 kg N/ha and year. However, N lost by runoff decreased to 74 kg N/ha and year. The N leaching losses and N runoff for scenario 2 were 106 and 90 kg N/ha and year, respectively.

In a real field, the readily available N in scenario 3 may be taken up by the crops during the vegetable season instead of being nitrified during the fallow, and thus lost by leaching when water percolates down the soil profile. According to K. Blombäck (pers. comm.), the rate of nitrification is very high at pH 6-7 in soil, as in Shiba Village, and there is no significant difference in crop uptake rate of NO_3^- and NH_4^+.

The P situation in scenario 3 was the same as in scenario 2, which is not strange, because the only P input was the manure, and the manure application rate was similar for scenarios 2 and 3.

Again, there is a quality aspect that must be considered if the fertiliser type is to be changed. Some vegetables, especially those belonging to the Chenopodiaceae botanical family such as spinach, beetroot and white beet, and some other species of different botanical families, for example lettuce, are able to hyperaccumulate NO_3^- for storage in the cell vacuoles during periods when the soil NO_3^- content is high. In Sweden, this is considered to be a crop product quality problem because of the health aspects of a high daily intake of NO_3^-. The metabolites of NO_3^- may decrease oxygen transport in humans and animals by binding to the haemoglobin in the red blood cells (homepage: National Food Administration, 2004-01-05). The NO_3^- may also be converted to nitrosamines in humans and other animals. The nitrosamines are considered to be potent carcinogens (Taiz and Zeiger, 1998). Thus, supplying N as NO_3^- might, in some cases, compromise vegetable quality and contribute to health hazards.

Scenario 4: Changing the crop rotation

In the fourth scenario, the crop rotation was changed by replacing cabbage with another crop. Radish replaced cabbage in the crop rotation in scenario 4a. Radish, however, was not included in the GLEAMS model database on crop characteristics, and all data used were for turnips. Thus, the results from the simulations are not very secure, but they can at least give an indication of the importance of the crop species grown in the system.

From the simulations of scenario 4a, one conclusion is that changing the crop rotation can be a way of further improving the nutrient management for the simulated field. Turnips increased the N use efficiency to 0.26 compared to 0.23 for the 'best management practice' in scenario 2. All losses were reduced, probably because of lower amounts of fertiliser input. Ammonia volatilisation was, for example, reduced by 30%, which was probably a result of reduced manure input. The nutrient losses may be even lower than shown by the results of the simulations since turnips have a shallower root system (30 cm) than radish (50 cm).

The P use efficiency increased to 0.107 in scenario 4a compared to 0.075 in scenario 2. According to the GLEAMS model database on crop characteristics, the N/P-ratio is lower in the harvest of turnips than in the harvest of cabbage. Thus, a larger amount of the P taken up by the turnips is removed from the field with the harvest. The results of the simulation showed that 63% of the P taken up by the crops in the cabbage-rotation was removed with the harvest, while the corresponding rate for the turnip-rotation was 72%. Furthermore, the P loss from runoff decreased to 3.6 kg P/ha and year, compared to 4.9 kg/ha and year in scenario 2.

Eggplant, which is both planted in Shiba village and included in the crop characteristics GLEAMS model database, would be another alternative in the crop rotation. It was used for the simulation in scenario 4b. Eggplant, however, did not improve the nutrient use efficiency in the case of N. One explanation may be that less of the N uptake by eggplant is found in the harvest than in cabbage. The ratio N yield/N uptake was calculated to be 53% in the case of eggplant, 62% for cabbage. When comparing the crop characteristics of eggplant and cabbage in the GLEAMS database, I also found that eggplant had a higher C/N ratio than cabbage. Thus, in similar biomass yields of eggplant and cabbage, the N content is lower in eggplant. However, the P use efficiency was increased by replacing cabbage with eggplant, despite a lower P yield/P uptake ratio of eggplant and an increase in P loss by runoff. This improved P use efficiency is probably achieved by lower input of P with the fertiliser (Table 7).

All N losses decreased in scenario 4b, especially in the case of N lost by leaching, compared to scenario 2. The average annual leakage of N was reduced by 25%, to 80 kg N/ha in scenario 4b compared to 106 kg N/ha in scenario 2. The N uptake rate was lower than in scenario 2, but higher than in scenario 4a, with turnips in the crop rotation. This may be explained by the deeper root system of eggplant, which has the ability to extracts nutrients from deeper soil layers.

No description of the time of the year when eggplant is grown in Shiba Village was available, but eggplant is actually a vegetable type that demands a warm climate to grow well. In Sweden, it can only be cultivated in greenhouses or on southern sites. Thus, the eggplant season would probably be in accordance with the time of the year when cabbage is grown in Shiba Village (Figure 3) since this period provides the warmest climate.

From a disease perspective, eggplant planted directly after pepper is not a good idea. Both eggplant and pepper are in the Solanaceae botanical family and generally, crops belonging to the same family are more susceptible to the same diseases and pests than crops of different botanical families (Agrios, 1997). In order to reduce the pesticide use in the very intensive-cropping systems in Shiba Village, it is of the utmost importance that the crop rotation is varied, and species susceptible to the same diseases and pests must not be planted next to each other. However, the small plots used for cropping pose a threat to achieving reduced pesticide use using a proper crop rotation since all crops are grown very close to each other. This facilitates the spread of diseases and pests.

Conclusions

The main conclusion from these simulations is that there is a great potential to improve the nutrient management in the investigated field in Shiba Village. The most effective way of doing it is to reduce the fertiliser input. Other management strategies, such as changing the crop rotation or changing fertiliser type, may contribute. There is also a need of more site specific data, for example, on crop yield and thus nutrient output in the harvest. Also, it is important to consider the K demand of the crop when developing strategies to reduce the N and P losses from the cropping system. However, to force the farmers to accept new ideas of management is not a good idea – it will probably not work in the long run. Therefore, the farmers must participate in the strategies planned to reduce the nutrient losses and they must make their own management decisions. The GLEAMS simulation model may be an important tool to examine which of these strategies are effective and which are not.

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