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Global and Regional Surface Nitrogen Balances in Intensive Agricultural Production Systems for the Period 1970–2030*1

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ABSTRACT

Global nitrogen (N) budgets for intensive agricultural systems were compiled for a 0.5 by 0.5 degree resolution. These budgets include N inputs (N fertilizer, animal manure, biological N fixation and atmospheric N deposition) and outputs (N removal from the field in harvested crops and grass and grass consumption by grazing animals, ammonia volatilization, denitrification and leaching). Data for the historical years 1970 and 1995 and a projection for 2030 were used to study changes in the recovery of N and the different loss terms for intensive agricultural systems. The results indicate that the overall system N recovery and fertilizer use efficiency slowly increased in the industrialized countries between 1970 and 1995, the values for developing countries have decreased in the same period. For the coming three decades our results indicate a rapid increase in both the industrialized and developing countries. High values of $> 80\%$ for fertilizer use efficiency may be related to surface N balance deficits, implying a depletion of soil N and loss of soil fertility. The projected intensification in most developing countries will cause a gradual shift from deficits to surpluses in the coming decades. The projected fast growth of crop and livestock production, and intensification and associated increase in fertilizer inputs will cause a major increase in the surface N balance surplus in the coming three decades. This implies increasing losses of N compounds to air (ammonia, nitrous oxide and nitric oxide), and groundwater and surface water (nitrate).

Key Words: animal manure, fertilizer, intensive agriculture, nitrogen, surface balance

INTRODUCTION

Human activities have accelerated the earth's nitrogen (N) cycle by increasing the natural rate of N fixation (Galloway *et al.*, 1995). Nitrogen fixation is the transformation of the highly abundant but biologically unavailable atmospheric dinitrogen (N_2) to "reactive" reduced and oxidized N forms such as ammonia (NH_3), nitrate (NO_3^-), nitrous oxide (N_2O) and nitric oxide (NO).

Biological N fixation occurs by specialized bacteria and algae, free-living or in a symbiotic relationship with higher plants, especially legumes. Estimates of biological N fixation in marine systems range from < 30 Tg year^{-1} to > 300 Tg year^{-1} (1 Tg = 10^{12} g) (Vitousek *et al.*, 1997). Biological N fixation in natural terrestrial ecosystems accounts for 100–290 Tg year^{-1} (Cleveland *et al.*, 1999). Further non-anthropogenic N fixation occurs during lightning (< 10 Tg year^{-1}) (Galloway *et al.*, 1995).

Anthropogenic N fixation occurs in N fertilizer (80 Tg year^{-1}) and energy (about 30 Tg year^{-1}) production and cultivation of leguminous crops in agricultural systems (40 Tg year^{-1}) (Galloway *et al.*, 1995). Hence, the total human-induced increase of global N fixation is about 150 Tg year^{-1}.

Apart from accelerating the global biogeochemical N cycle, human activities have also increased the mobility of reactive N within and between terrestrial and aquatic ecosystems and the atmosphere (Vitousek *et al.*, 1997). While most of the reactive N emissions occur locally in the terrestrial system, the influence of these emissions spreads regionally and globally as it moves through water and air across political and geographic boundaries.

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The accelerated N cycling has consequences for (agro-)ecosystems, groundwater and surface water systems. Atmospheric N deposition rates onto the earth's surface have increased between 3- and more than 10-fold since pre-industrial times (Galloway *et al.*, 1995). In terrestrial ecosystems eutrophication by increased N deposition has a number of important negative impacts, including i) shifts in plant species composition towards nitrophilic species (Bobbink *et al.*, 1998); ii) toxicity of N gases and aerosols to some species, especially near sources of NH_x and NO_y; iii) increased susceptibility of plants to secondary stress factors (Brown, 1995); iv) enhanced nitrate (NO_3^-) leaching; and, v) emission of nitrous oxide (N_2O) and nitric oxide (NO) from denitrification and nitrification in soils (Mosier *et al.*, 1998). A possible positive impact is enhanced carbon storage in biomass and soil organic matter by N deposition (Holland *et al.*, 1997; Nadelhoffer *et al.*, 1999) which may counteract the increasing atmospheric CO_2 concentration.

Increased concentrations of nitrate have been observed in groundwater in many (primarily agricultural) regions, although the magnitude of this increase is difficult to determine outside a few well-characterized aquifers. High levels of nitrate in drinking water have given cause for concern in some industrialized regions, *e.g.*, Western Europe. Recently, nitrate concentrations in groundwater have also been shown to increase in developing countries with intensive agricultural production (Singh *et al.*, 1995).

Nitrogen concentrations in many rivers have increased over time (Heathwaite, 1993; Johnes and Burt, 1993). Howarth *et al.* (1996) estimated that riverine N fluxes from most of the temperate regions surrounding the North Atlantic Ocean had increased from pre-industrial times from 2- to 20-fold, while for the North Sea region this increase has been 6- to 20-fold. With increasing population densities and agricultural production riverine N fluxes are also increasing in many developing countries. In aquatic systems typical consequences of eutrophication by increased N loads include elevated primary production and respiration. In its most serious manifestations aquatic eutrophication leads to algal blooms, algal scum, enhanced benthic algal growth and massive growth of submersed and floating macrophytes, changes in food webs and species composition, and depletion of the oxygen reserve, with consequences for water quality and harvestable fisheries (EEA, 2001; Hallegraeff, 1993; Turner and Rabelais, 1994; Vollenweider *et al.*, 1992).

Agricultural production systems have been identified as the major cause of increased reactive N emissions to the atmosphere (Bouwman *et al.*, 1997; Mosier *et al.*, 1998) and groundwater and surface water systems (Van Drecht *et al.*, 2003). Projections indicate that the world population may increase from about 6 now to 8.2 to 9.3 billion inhabitants in 2030 (Nakicenovic *et al.*, 2000). Food production will have to increase to meet the increasing demand for the growing population, while with increasing prosperity and falling production costs dietary patterns may shift towards a higher share of meat and milk. This will inevitably lead to a fast increase in fertilizer use and the volume of animal manure produced in livestock production systems.

Often farm-gate or whole system budgets or balances are made, which consider the N input, output and total loss for the farm or system considered without specifying where and in which form losses occur (Van der Hoek and Bouwman, 1999). Other studies use a lumped approach to estimate country surface N balance surpluses without specifying the fate of the surplus N (Hansen, 2000; OECD, 2001; Sheldrick *et al.*, 2002; Smaling *et al.*, 1993).

In this study we focus on the geographic distribution of the fate of nitrogen in the environment. Therefore, the surface N balance approach considering all relevant input, output and loss terms for a given land area is more appropriate than farm-gate or system balances. We use geographically distributed N budgets for agro-ecosystems compiled for a 0.5 by 0.5 degree resolution within the Integrated Model to Assess the Global Environment (IMAGE) (IMAGE-team, 2001). We present an inventory of the surface N balance input terms for all countries of the world, the recovery of N in harvested crops, cut grass and hay, grass consumed by grazing animals, and N losses to air, groundwater and surface water via NH_3 volatilization, denitrification and leaching. We describe the situation in 1970 and the mid-1990s. The projection for world agriculture presented by the Food and Agriculture Organization (FAO) of the United Nations (Bruinsma, 2003) is used to calculate N balances for the period up till 2030.

An earlier version of our surface N balance approach for the situation in the mid-1990s is described by Van Drecht *et al.* (2003). The approach for a number of input and output terms was updated as described below. The results of this study will be used to assess the fate of N in the hydrological system up to the river mouths for the 1970–2030 period (Bouwman *et al.*, 2005a).

In this paper we subsequently discuss the methods used to calculate the surface N balance, present the results for different countries and world regions. Finally, we summarize our conclusions.

METHODS AND DATA USED

Land cover

The basis of the surface N balance calculations is the land cover distribution. Data for 1970–1995 from FAOSTAT (FAO, 2001) and 2030 from Bruinsma (2003) were implemented in the IMAGE model (IMAGE-team, 2001) by Bouwman *et al.* (2005b) to generate 0.5 by 0.5 degree global land cover maps giving the grid cells covered by agriculture and natural ecosystems. Each agricultural grid cell consists of fractions of grassland, rice, temperate cereals, maize, tropical cereals, pulses, root and tuber crops, oil crops and other crops. These categories were aggregated to four broad groups, including grassland, wetland rice, leguminous crops (pulses, soybeans) and other upland crops (Fig. 1). Wetland rice is distinguished because of the characteristic conditions in inundated fields important for NH_3 volatilization (Bouwman *et al.*, 2002b), leaching and denitrification (Singh *et al.*, 1995). Van Drecht *et al.* (2003) lumped the leguminous crops with the other upland crops. Here, leguminous crops form a separate group because of their important role in the agricultural N cycle. Leguminous crops can fix atmospheric N through symbiosis with rhizobium bacteria in their root nodules and generally receive only low N fertilizer inputs as a starter.

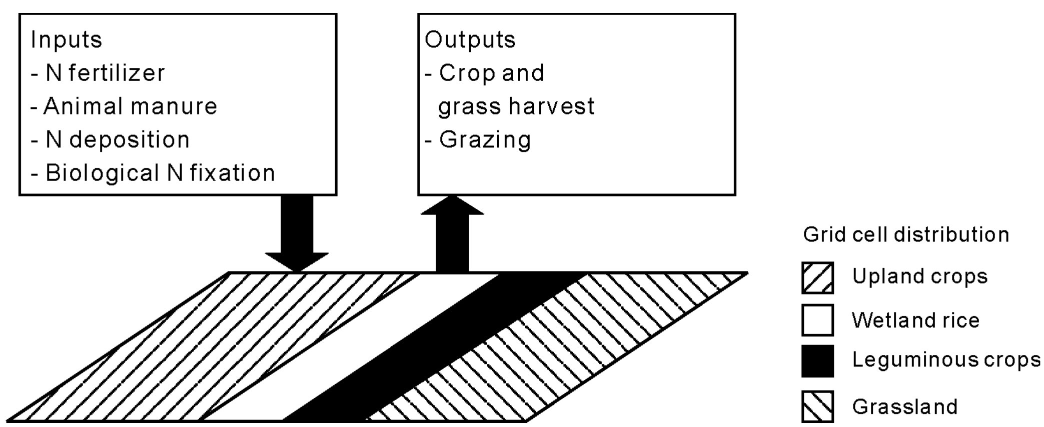

Fig. 1 Surface N balancing for agricultural systems within each 0.5 by 0.5 degree grid cell. Balances are made for fractions of grid cells covered by grassland, wetland rice, leguminous crops and other upland crops. The surface N balance is the difference between the sum of all inputs and the N removal from the field by crop and grass harvesting and grass consumption by animals. Any surface N balance surplus is subject to NH_3 volatilization, denitrification or leaching.

Within the areas of grassland, we distinguish between semi-natural, pastoral and mixed + landless livestock production systems according to Bouwman *et al.* (2005b) who used recent estimates from Seré and Steinfeld (1996). Areas of grassland receiving synthetic N fertilizers (from IFA/IFDC/FAO (2003)) form a subset of the grassland in mixed + landless farming systems. Since there is no projection available for grassland from Bruinsma (2003), we use the projection presented by Bouwman *et al.* (2005b) with decreasing areas of grassland in mixed + landless systems in most industrialized countries, and nearly constant to increasing areas in most developing countries (Table I).

TABLE I

Areas of grassland in mixed + landless livestock production systems and arable land in 9 world regions for 1970, 1995 and 2030[a)]

	1970		1995		2030	
	Grassland	Arable land	Grassland	Arable land	Grassland	Arable land
	Mha					
North America	171	234	170	224	127	251
Western Europe[b)]	68	96	60	86	53	76
Transition countries[c)]	100	282	90	266	96	273
Latin America	47	116	66	160	104	200
Middle East and North Africa[d)]	21	80	25	91	26	94
Sub-Saharan Africa[e)]	41	142	49	167	61	217
South Asia[f)]	14	205	12	213	22	220
East Asia[g)]	72	130	75	142	70	130
Southeast Asia[h)]	13	69	12	88	15	102
World	551	1405	565	1494	584	1609

[a)] Data from Bouwman *et al.* (2005b). The projection for 2030 is based on Bruinsma (2003).
[b)] Austria, Belgium, Denmark, Finland, France, Germany, Gibraltar, Greece, Iceland, Ireland, Italy, Luxembourg, Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, and United Kingdom.
[c)] Countries of former USSR, Albania, Bosnia and Herzegovina, Bulgaria, Croatia, Czech Republic, Hungary, Macedonia, Poland, Romania, Slovakia, Slovenia, Yugoslavia.
[d)] Algeria, Bahrain, Cyprus, Egypt, Iran, Iraq, Israel, Jordan, Kuwait, Lebanon, Libya, Morocco, Oman, Qatar, Saudi Arabia, Syria, Tunisia, Turkey, United Arab Emirates, Western Sahara and Yemen.
[e)] Angola, Benin, Botswana, Burkina Faso, Burundi, Cameroon, Cape Verde, Central African Republic, Chad, Comoros, Democratic Republic of Congo, Republic of Congo, Côte d'Ivoire, Djibouti, Eritrea, Ethiopia, Equatorial Guinea, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Kenya, Lesotho, Liberia, Madagascar, Malawi, Mali, Mauritania, Mauritius, Mayotte, Mozambique, Namibia, Niger, Nigeria, Réunion, Rwanda, Saint Helena, Sao Tome and Principe, Senegal, Seychelles, Sierra Leone, Somalia, South Africa, Sudan, Swaziland, Tanzania, Togo, Uganda, Zambia and Zimbabwe.
[f)] Afghanistan, Bangladesh, Bhutan, India, Nepal, Pakistan and Sri Lanka.
[g)] China, Democratic People's Republic of Korea, Republic of Korea and Mongolia.
[h)] Brunei Darussalam, Cambodia, Indonesia, Laos, Malaysia, Myanmar, Philippines, Singapore, Thailand, East Timor and Viet Nam.

It should be noted that the FAOSTAT (FAO, 2001) land-use data for 1970–1995 used in the IMAGE model do not include the corrections for crop yields made by Bruinsma (2003). This implies that the areas of agricultural land use generated for 2030 by the IMAGE model are smaller than those projected by Bruinsma (2003), particularly for Sub-Saharan Africa and Latin America. The estimated changes in land use are, however, consistent. The projected fast expansion of arable land between 1995 and 2030 (Table I) is caused by major increases in crop production in primarily the developing countries (Bouwman *et al.*, 2005b; Bruinsma, 2003). A large part of the growth of crop production is associated with the increasing demand for feed concentrates for livestock production (Bouwman *et al.*, 2005b).

The 0.5 by 0.5 degree resolution maps for the different agricultural systems were used to allocate all surface N balance input and output terms to fractions of grid cells covered by wetland rice, leguminous crops, other upland crops or fertilized grassland (Fig. 1).

Surface N balance

We use the concept of the annual surface N balance (Fig. 1), which includes the N inputs and outputs for a given area of land, here 0.5 by 0.5 degree grid cells. N inputs include biological N fixation (N_{fix}), atmospheric N deposition (N_{dep}), application of synthetic N fertilizer (referred to as fertilizer, N_{fert}), and animal manure and animal N excreted during grazing (N_{man}). Outputs in the surface N balance include N removal from the field by crop harvesting, cutting of hay and grass and grass consumption by grazing animals. Hereinafter, this N removal is referred to as crop N export (N_{exp}).

The remainder surface N balance surplus (N_{sur}) i ssubject to NH_3 volatilization (N_{vol}), denitrification (N_{den}) or leaching and surface runoff (N_{lea}). The surface N balancing is a steady state approach, not taking into account change of N in soil organic matter:

$$N_{\mathrm{sur}} = N_{\mathrm{fix}} + N_{\mathrm{dep}} + N_{\mathrm{fert}} + N_{\mathrm{man}} - N_{\mathrm{exp}} \tag{1}$$

where the potential loss to groundwater and surface water N_{pot} is calculated as the difference between N_{sur} and N_{vol}:

$$N_{\mathrm{pot}} = N_{\mathrm{sur}} - N_{\mathrm{vol}} \tag{2}$$

Leaching is defined as the loss of nitrate in percolating water from the root zone to the subsoil where it is no longer available to plant roots. Denitrification in soil is calculated as an empirical fraction f_{den} of N_{pot}:

$$N_{\mathrm{den}} = f_{\mathrm{den}} N_{\mathrm{pot}} \tag{3}$$

The fraction f_{den} is calculated with a model that combines the effect of temperature, crop type and soil and hydrological conditions (Van Drecht *et al.*, 2003). Nitrate leaching is then calculated as follows:

$$N_{\mathrm{lea}} = N_{\mathrm{pot}} - N_{\mathrm{den}} = (1 - f_{\mathrm{den}}) N_{\mathrm{pot}} \tag{4}$$

It should be noted that N_{lea} includes surface runoff and drainage. Surface runoff is particularly important in sloping regions where soil material containing nutrients is lost through erosion. It is difficult to quantify the actual N loss through erosion, but at larger scales this process is less important than at the plot scale, since much of the eroded soil is redeposited at the foot of slopes (Robinson, 1979). Further N losses occur through drainage systems in poorly drained or irrigated fields, or as a result of temporary drainage of wetland rice fields. The consequence of our approach is that nitrate leaching might be slightly overestimated is such cases. It should also be noted that during transport of nitrate in groundwater and surface water there is also denitrification, storage and retention as discussed in Van Drecht *et al.* (2003).

Many countries have vast areas of extensively used grassland, typically with small N inputs. Contrary, other countries have primarily mixed and landless systems characterized by much larger N inputs. To make a good comparison between different countries, we exclude areas of pastoral, semi-natural and marginal grassland distinguished by Bouwman *et al.* (2005b), and consider the N balances for crop and mixed + landless livestock production systems together. Including pastoral systems would give an unrealistic impression of the intensity of agricultural systems.

We modified the approach of Van Drecht *et al.* (2003) by considering the different crops within each grid cell to represent a rotating system, where the surplus of one crop may serve to compensate for the deficit of the next crop. Where rice occurs in a rotating crop system, we assume that denitrification occurs within the rice growing season, and any surplus thereafter is passed on to the next crop.

Biological N fixation

Biological N fixation by pulses and soybeans is calculated from crop production data (FAO, 2001) and N content (Tables II and III). Total biological N fixation in biomass during the growing season of pulses and soybeans is calculated by multiplying the N in the harvested product by a factor of 2 to account for all above and below-ground plant parts, according to Mosier *et al.* (1998). Furthermore, we use a rate of non-symbiotic biological N fixation of 5 kg ha^{-1} year^{-1} for non-leguminous upland crops and grassland and 25 kg ha^{-1} year^{-1} for wetland rice as proposed by Smil (1999).

TABLE II

Production of upland crops, leguminous crops and wetland rice for 1970, 1995 and 2030 for world regions

Region	1970			1995			2030		
	Upland crops	Leguminous crops	Rice	Upland crops	Leguminous crops	Rice	Upland crops	Leguminous crops	Rice
	Mt year^{-1}								
North America	377	33	2	568	70	4	835	135	5
Western Europe	386	2	1	495	6	2	567	8	2
Transition countries	559	9	1	478	6	1	563	5	1
Latin America	491	7	3	901	46	7	1389	133	12
Middle East and North Africa	119	2	4	264	4	7	443	7	13
Sub-Saharan Africa	235	4	2	408	7	6	827	20	16
South Asia	426	13	67	842	20	120	1615	41	213
East Asia	455	16	102	1037	20	171	1603	40	194
Southeast Asia	201	1	47	522	5	100	765	10	155
Oceania	45	0	0	83	2	0	118	3	1
Japan	59	0	10	50	0	8	39	0	6
World	3355	86	239	5647	186	427	8763	403	618

TABLE III

N contents of harvested parts for 34 crops and crop groups distinguished in this study

Crop/crop group	N content	Crop/crop group	N content
	g kg^{-1}		g kg^{-1}
Wheat	19	Citrus fruit	1
Paddy rice	13	Fruit excluding melons	1
Maize	14	Cocoa beans	14
Barley	17	Rapeseed	35
Millet	15	Oil palm fruit	15
Sorghum	15	Soybeans	35
Other cereals	16	Groundnuts in shell	40
Potatoes	3	Sunflower seed	34
Sweet potatoes	3	Sesame seed	33
Cassava	2	Other oilseeds	30
Other root crops	3	Coconuts	0
Plantains	2	Coffee, green	24
Sugar beets	2	Tea	78
Sugar cane	2	Tobacco leaves	3
Pulses, total	35	Seed cotton	29
Vegetables and melons	2	Fibre crops primary	81
Bananas	2	Rubber	0

Use of N fertilizers

Mean country N fertilizer application rates for the mid-1990s for wetland rice, leguminous crops and fertilized grassland are, with a few exceptions, taken directly from IFA/IFDC/FAO (2003). The N fertilizer application rates for other upland crops are calculated as the difference between total country N fertilizer use (IFA, 2002) and the sum of the N use in wetland rice, leguminous crops and grassland (IFA/IFDC/FAO, 2003) divided by the area of other upland crops for each country.

For 1970 the application rates are multiplied with the change in crop yield compared to 1995 (Bruinsma, 2003) ignoring changes in crop N recovery. For 2030 we assumed that fertilizer inputs increase by 80%–90% of the crop yield increase to achieve consistency with the original projection of Bruinsma (2003), implying an increase in the fertilizer N recovery.

For 1970 and 1995 we use the mix of fertilizers presented by IFA (2002). For 2030 we assume that the 2001 mix of fertilizers provided by IFA (2002) will not change, except for the use of ammonium bicarbonate in China. We assumed that 90% of the use of the latter fertilizer type will be substituted by urea in 2030.

Management of animal manure

N excretion rates per head for large ruminants (dairy and nondairy cattle, buffaloes), small ruminants (sheep and goats), pigs, poultry, horses, asses, mules and camels from Van der Hoek (1998) are used for 1970, 1995 and 2030 (Fig. 2), with excretion per unit of product decreasing along with animal productivity. The number of animals and production for each category in mixed + landless systems are taken from Seré and Steinfeld (1996) (Fig. 2). Pigs and poultry are assumed to occur in mixed + landless systems only.

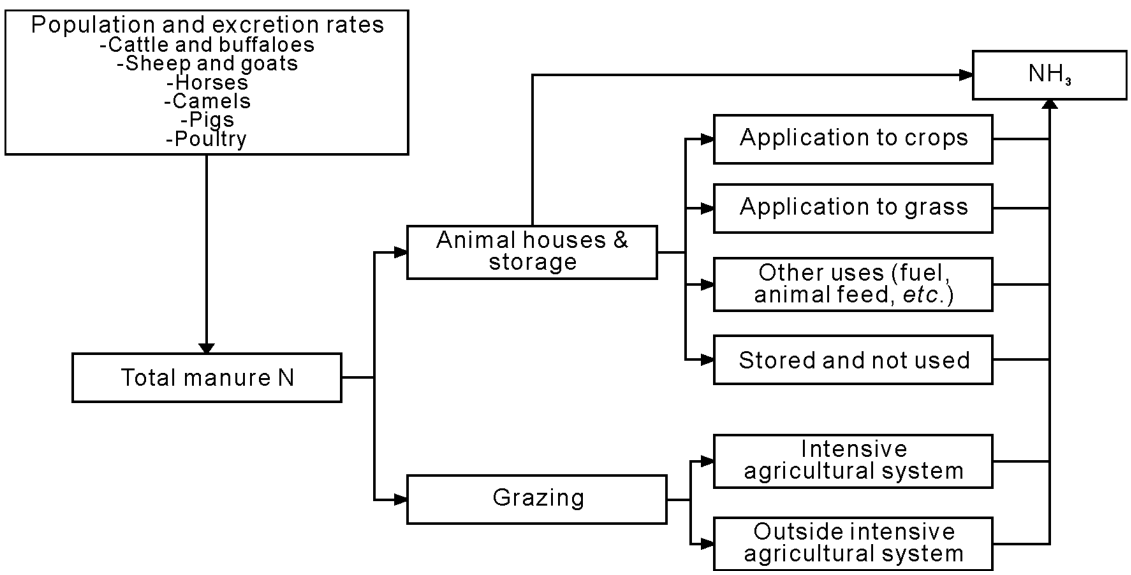

Fig. 2 Scheme representing the distribution of N from animal manure, animal houses and storage, application to cropland and grassland, and other uses, grazing and NH_3 volatilization in mixed + landless (intensive) systems. Animal manure used as fuel or stored but unused manure, and excretion outside the intensive agricultural system is considered not to be part of the agricultural system.

Within each system, part of the animal manure is excreted during grazing, and the remainder is stored or collected (Fig. 2). All manure produced by pigs and poultry is assumed to be stored or collected. The fraction of the manure that is excreted by ruminants in pastures is calculated from the fraction of grass in the ration as discussed by Bouwman *et al.* (2005b). For grazing animals the fraction of the animal manure that is stored or collected in farm houses and storage systems depends on the country considered. Animal manure N available for application to crops and grassland is all stored or collected manure, excluding i) manure excreted outside the agricultural system, for example in urban areas, forests, roadsides, *etc.*; ii) stored manure that is not used, such as in many lagoon systems in North America; iii) manure used as a fuel or for other purposes according to Mosier *et al.* (1998); and vi) NH_3 volatilization from storage systems calculated according to Bouwman *et al.* (1997) (Fig. 2). For India we assume that a large part of the total amount of animal manure is excreted in forests, during roadside grazing or scavenging in villages and urban areas, and is not part of the agricultural system, based on Van der Hoek (2001). As a consequence about 9 out of 15 Tg of N is not included in intensive agricultural systems in India for 1995. For East Asia we assume that the fraction of animal manure that is not recycled increased between 1970 and 1995 from about 10% to 20%, and this increase will continue in the coming three decades to about 30%.

We assumed that 50% of the stored and available animal manure is applied to arable land and the remainder to grassland (Lee *et al.*, 1997). In most developing countries 95% of the available manure is assumed to be applied to crops and 5% to grassland, thus accounting for stubble grazing in croplands, and the lower importance of grass compared to crops in developing countries (Seré and Steinfeld, 1996).

For European Union (EU) countries with high animal manure inputs we use maximum application rates of 170–250 kg N ha^{-1} $year^{-1}$ for 2030 based on existing EU policies, assuming that this is consistent with the projections for livestock production (Bruinsma, 2003) on the basis of currently decreasing N excretion per unit of product. This has consequences for only a few countries, since in most EU countries the average manure application rate is below the prescribed maximum in 2030.

Because of lack of data we do not consider the use of human excrements ("night soil") to fertilize agricultural fields. Furthermore we ignore the use of sewage sludge. In developing countries this source of N is probably not important due to the low degree of wastewater collection and treatment, while in industrialized countries its use may be restricted due to environmental regulations related to pathogens and heavy metals in sludges.

Atmospheric N deposition

Atmospheric N deposition rates for the mid-1990s were calculated with the STOCHEM global chemistry-transport model (Collins *et al.*, 1997), which was run with gridded inventories for energy-related, biomass burning, agricultural and natural emissions of nitrogen oxides and NH_3 as described by Bouwman *et al.* (2002d). This model simulates long-range N transport and deposition. We calculated short-range dry deposition of NH_3 as 50% of the NH_3 emissions taken from Bouwman *et al.* (1997) assuming re-deposition within 0.5 by 0.5 degree grid cells (Bouwman *et al.*, 2002d). Atmospheric deposition fields cover all agricultural and natural ecosystems, showing considerable spatial variability with highest rates occurring in South and East Asia and low rates in remote regions.

To obtain estimates for 1970 and 2030 the deposition fields for the mid-1990s are scaled using emission inventories for N gases for 1970 (Olivier *et al.*, 2001) and scenario calculations for 2030 for IPCC's Special Report on Emission Scenarios-B2 scenario from Nakicenovic *et al.* (2000) which was implemented with the IMAGE model (IMAGE-team, 2001). The basic assumptions on population and economic growth of this scenario are very similar to those of the FAO projection (Bruinsma, 2003).

Crop N export

Crop N export includes the removal of N from i) arable land in harvested crop parts, and ii) grassland in cut grass and hay and grass consumed by grazing animals. The removal of N from arable fields in the harvested product is calculated from crop production data (FAO, 2001) and projections and N contents for 34 crops used by Bruinsma (2003) (Tables II and III). This is an improvement of the approach of Van Drecht *et al.* (2003) who used 8 broad crop groups to calculate the N removal in harvested parts. We aggregate the crop N export to the broad categories wetland rice, leguminous crops and other upland crops.

The FAOSTAT data (FAO, 2001) lack information on the production of fodder crops (such as fodder maize and beet, alfalfa) which are generally produced for on-farm use and are not marketed. This problem was recognized by Van Drecht *et al.* (2003). In this study we therefore use data from OECD (2001) on fodder crop production and N content for OECD (Organization for Economic Co-operation and Development) countries and add the N removal in fodder crops to our estimate for crop N removal for 1995. For 1970 and 2030 we assume that the absolute amount of N in harvested fodder crops in the OECD countries is proportional to total meat production.

For the non-OECD countries the contribution of fodder crops to total crop N removal is assumed to be negligible. This assumption is based on the reported areas used for fodder production, which make up less than 4% of the total arable area in developing countries and will not increase in the coming decades (Bruinsma, 2003).

The N input from crop residues that remain in the field is ignored, since this is assumed to be an internal cycle with no effects on the annual surface N balance. We ignore agricultural waste burning and the use of crop residues (as fuel, animal feed or for other purposes), which leads to an overestimation of denitrification and leaching losses in our approach. Ignoring burning of residues has no effect on the estimates for the system N recovery or fertilizer use efficiency. Contrary, not accounting for the use of residues causes an underestimation of crop N export. However, for most crops the amount of N in above-ground crop residues is much smaller than that in the harvested parts, particularly for cereals, and the error caused by ignoring this term is probably small compared to the uncertainty in the estimates of N removal in crop N export.

Data on grass production and consumption by grazing animals and cutting of grass and hay are not readily available. We therefore calculate N removed from the field in harvested grass and grass consumption by grazing animals as 0.6 times the sum of the N inputs (Fig. 1) minus the NH_3 emission. This means that N consumption exceeded 60% of the N inputs from fertilizer and animal manure. This is based on the higher efficiency of N uptake as a result of the longer growing period of grass compared to crops, which have N use efficiencies of 40%–50% of the N input from fertilizers (Van der Hoek and Bouwman, 1999).

Ammonia volatilization

In the surface balance approach the manure applied to agricultural land excludes NH_3 volatilization from animal manure stored or collected in animal houses, feedlots and other storage systems. Ammonia volatilization from the application of animal manure and N fertilizers is calculated for each 0.5 by 0.5 degree grid cell with a residual maximum likelihood approach based on a data set of about 1 700 measurements (Bouwman *et al.*, 2002b). Calculated NH_3 volatilization rates are based on factors related to agricultural management (crop type, fertilizer type, and fertilizer application technique) and factors related to environmental conditions (climate, soil pH, and CEC). The resulting mean NH_3 volatilization from animal manure and N fertilizers is 23% and 14%, respectively, of the N applied, with large differences between fertilizer types and spatial variability within areas of wetland rice, leguminous crops, other upland crops and grassland (Bouwman *et al.*, 2002b). Ammonia volatilization from N excretion by grazing animals (Fig. 2) is calculated on the basis of Bouwman *et al.* (1997).

Denitrification in the root zone and nitrate leaching to groundwater

We assume that inputs of all reduced N compounds not taken up by plant roots or lost by NH_3 volatilization, will be nitrified forming nitrate in soils. Nitrate is subject to denitrification, and since it is highly mobile in soils, is leached during periods with excess precipitation (Di and Cameron, 2002). Soil denitrification occurs in or just below the root zone when temperature is above freezing point and soil water content is high and oxygen availability low, forming N_2, N_2O and NO (Firestone and Davidson, 1989). Estimates of emissions of N_2O and NO from agricultural systems, based on a similar approach for estimating N inputs from fertilizers and animal manure as discussed above, are presented elsewhere (Bouwman *et al.*, 2002a; Bouwman *et al.*, 2002c).

For rainfed crops we use a model that combines the effect of temperature, crop type, soil properties and hydrological conditions on annual mean nitrate leaching and denitrification rates as described by Van Drecht *et al.* (2003), relying on simplifications of empirical models (Kolenbrander, 1981; Kragt *et al.*, 1990; Simmelsgaard *et al.*, 2000). According to this model climates with large annual precipitation excess have high leaching rates and short residence times of nitrate in the soil solution. In dry climates the annual water flow through the soil is small and the residence time of nitrate is longer than in humid climates. N transformation rates are higher in tropical than in temperate climates.

Locally in dry climates with a short rainy season soils may have a 'dead dry' horizon, which receives neither percolation water from above nor capillary rise from below. In this horizon there may be accumulation of salts including carbonates or salts of nitrate. However, our approach does not account for such processes (Van Drecht *et al.*, 2003).

In the model soil texture, drainage class and organic C content influence denitrification through the soil's water and oxygen status. In fine textured soils and soils with poor drainage, anaerobic conditions favoring denitrification are more easily reached and maintained for longer periods than in coarse-textured and well-drained ones. Soil organic carbon content is used as a proxy for the carbon supply for denitrification.

This approach for rainfed crops does not account for irrigation. The conditions for leaching and denitrification in irrigated systems differ from those in rainfed systems. In addition, N fertilizer application rates are generally higher in irrigated than in rainfed systems (Singh *et al.*, 1995). Data on irrigation water use are not available at the scale of our calculations. Therefore, irrigation is ignored in all crop areas except wetland rice, and this may lead to errors in the estimation of nitrate leaching rates in about 10% of the global arable area (Van Drecht *et al.*, 2003).

In wetland rice systems the effects of soil texture, soil organic carbon, drainage and precipitation surplus are ignored, because these systems were assumed to be predominantly anaerobic. Since rice is primarily produced in subtropical and tropical climates, denitrification in wet rice fields is assumed to be 75% of the surface N balance surplus, so that the complement (25%) is leached from the root zone. This is based on measurements indicating that about 30% of the total N input is lost by denitrification (Van Drecht *et al.*, 2003).

System N recovery and fertilizer use efficiency

Livestock and crop production in mixed systems can be considered as one system, because the by-products of one activity (crop by-products, crop residues, manure) often serve as inputs for another. We therefore calculate the overall system N recovery for arable land and grassland occurring in intensive agricultural systems, which is the crop N export (sum of the N in harvested crop and grass parts and grass consumption by grazing animals), expressed as a percentage of the sum of all N inputs.

In addition, we calculate the nitrogen use efficiency (NUE) for crops only. NUE is the N in harvested crop parts expressed as a percentage of the N fertilizer and animal manure inputs. In the calculation of NUE we exclude N fixing leguminous crops, which generally receive small doses of N fertilizer (IFA/IFDC/FAO, 2003). For these crops NUE is not a meaningful concept. The fertilizer input includes inputs of animal manure N where applicable, since in many parts of the world this forms a major contribution to total inputs in mixed systems.

RESULTS AND DISCUSSION

All calculations are done for each of the 0.5 by 0.5 degree grid cells with intensive agricultural systems, but for presentation in this paper the results are aggregated to the scale of countries and world regions. It should be noted that much of the spatial variability at the 0.5 by 0.5 degree resolution is lost due to aggregation.

Western European countries in 1995

The results for the mid-1990s show considerable differences in the total inputs and the relative importance of the different input and output terms between individual Western European countries (Fig. 3). There is a clear difference in total inputs between the Mediterranean countries (Greece, Italy, and Spain) and northern European countries (The Netherlands, Belgium, Denmark, Ireland, and UK). Fertilizer N inputs also vary widely, with average application rates of less than 40 kg ha^{-1} year^{-1} in the Mediterranean and middle European countries (Spain, Portugal, Greece, Switzerland and Austria) and very high rates in most northern European countries (Belgium, Ireland, Denmark, Germany, and The Netherlands).

Some of the countries are characterized by intensive livestock production and have large absolute

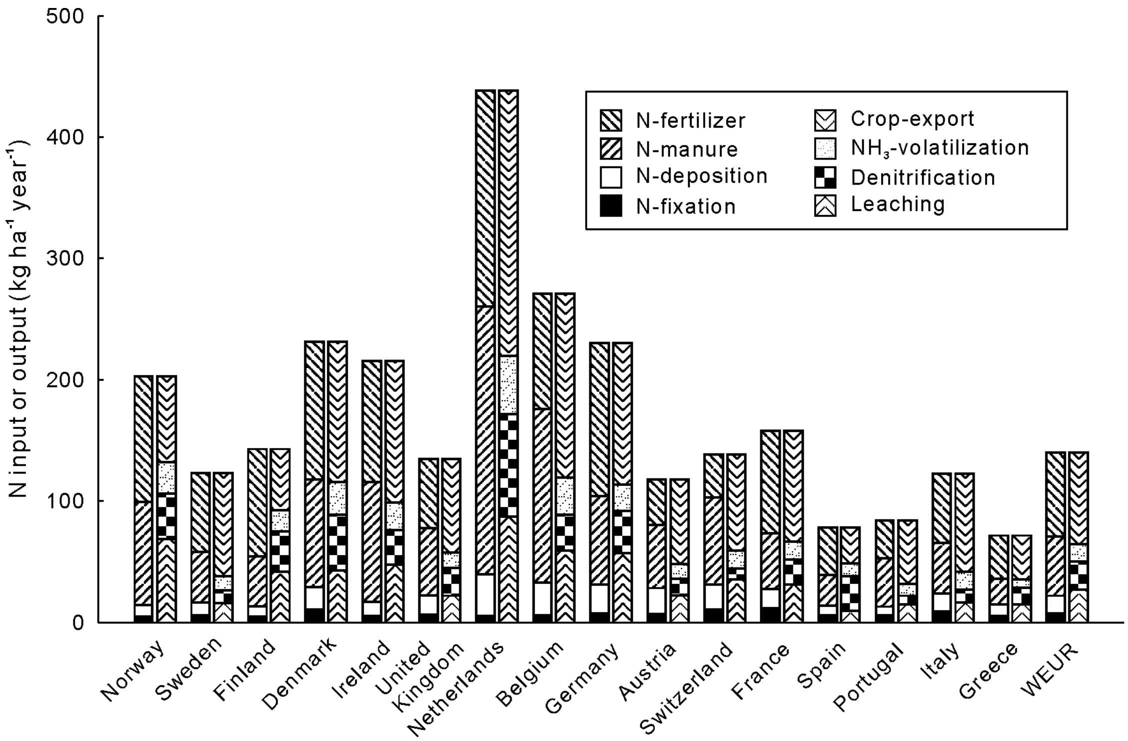

Fig. 3 Surface N balance terms for 1995 for Western European (WEUR) countries. For each country the left-hand bar indicates the N inputs, and the right-hand bar the N outputs. The crop export includes N removal from the field in harvested crops and grass cut or grazed by animals. N-manure includes both N in manure applied to crops and grassland and N excreted during grazing, and excludes NH_3 volatilization from animal houses and stored manure. NH_3 volatilization is the NH_3 volatilization from application of N fertilizer and animal manure and from N excreted by grazing animals. Leaching of nitrate is the nitrate that percolates to the subsoil below the rooting zone. During transport to the surface water or groundwater system further denitrification may occur. The N balances were calculated for areas of arable land and grassland in mixed systems (Table I), excluding pastoral systems, (semi-)natural and marginal grassland.

and relative inputs of N from animal manure (such as The Netherlands and Belgium where N-manure is about 50% of the total input). N inputs from animal manure vary from less than 30 kg ha^{-1} year^{-1} (Greece, Spain) to more than 200 kg ha^{-1} year^{-1} (The Netherlands). It is not surprising that the mean N deposition rates are closely correlated with the animal manure N inputs. In all Western European countries the input from biological N fixation is small compared to the other input terms.

The overall system N recovery ranges from about 35% of the total N input (Norway and Finland) to close to 60% (*e.g.*, U.K. and Sweden), while the mean for Western Europe is about 50%. These differences are due to various factors, such as climate, the mix of crops, their N content and yields, importance of grazing versus confined animals, availability and use of animal manure, occurrence of excessive use of animal manure, and the importance of biological N fixation by leguminous crops.

While the N remaining for denitrification and leaching is the complement of crop and grass N removal and NH_3 volatilization, the importance of N leaching losses relative to denitrification varies as a function of climate and soil conditions. Regarding climate, our model predicts low leaching rates for countries with Mediterranean climates (*e.g.*, Italy, 18% of total inputs; Portugal, 17%; Spain, 12%; Greece, 21%), and much higher rates in countries with humid climates (*e.g.*, Germany, 29%; Finland, 29%, Sweden, 34%). The influence of soil conditions is clear where, for example, agricultural land is found on peat or heavy clay soils with shallow water tables, as in The Netherlands. Such conditions are prone to high denitrification losses, and as a consequence have low leaching rates.

It is difficult to validate the modeled nitrate leaching from the root zone for 0.5 degree grid cells with point measurements in groundwater. However, comparison of the computed nitrate concentrations in the shallow groundwater layer (data not shown) with other estimates (RIVM/RIZA, 1991) gives a good agreement.

World regions for 1970–2030

In comparing world regions for 1995 (Fig. 4), it is obvious that there are large differences between systems dominated by irrigated agriculture with more than one crop per year (South Asia, East Asia) and rainfed systems with one crop per year (*e.g.*, Western Europe). The average fertilizer N application rates are low in most of Africa, Latin America and the countries of the former USSR. The N fertilizer use in the former USSR has sharply dropped since 1990 from much higher levels in the mid-1980s.

In many world regions intensification of agricultural production has been fast in the past decades (Fig. 4), and this development will continue at a somewhat slower rate in the coming decades. The development of N fertilizer use for the period 1970–1995 and the projection for 2030 shows that fertilizer use will continue to increase considerably in most, particularly Asian, regions (Table IV and Fig. 4).

TABLE IV

Nitrogen fertilizer use for 1970, 1995 and 2030[a)]

Region	1970	1995	2030
	Mt $year^{-1}$		
North America	7.7	13.2	18.2
Western Europe	6.7	10.4	10.7
Transition countries	7.2	4.7	5.9
Latin America	1.2	5.0	7.6
North Africa + Middle East	0.8	3.5	5.0
Sub-Saharan Africa	0.4	1.0	1.9
South Asia	1.9	14.2	20.1
East Asia	3.7	24.4	32.6
Southeast Asia	0.6	4.4	5.9
World	31	83	110

[a)] The projection for the year 2030 is based on Bruinsma (2003).

Current N inputs from animal manure vary from less than 10 kg ha^{-1} $year^{-1}$ to more than 50 kg ha^{-1} $year^{-1}$, and are the highest in Western Europe and East Asia and much lower in world regions dominated by crop production systems and with less intensive livestock production. Inputs from animal manure will grow in all world regions in the coming three decades, except for Western Europe (Fig. 4). Within the latter region EU regulations for animal manure N inputs will be bound to a maximum rate, and this has consequences in some countries with current high inputs, such as The Netherlands and Denmark. Atmospheric N deposition is generally the highest in world regions with intensive livestock production.

Global total N inputs from fertilizers and animal manure in intensive agricultural systems have almost doubled between 1970 and 1995 from about 74 Tg $year^{-1}$ to 138 Tg $year^{-1}$, whereby N manure contributed 44 Tg $year^{-1}$ or about 60% in 1970 and 55 Tg $year^{-1}$ (40%) in 1995. For 2030 we estimated a total N input from fertilizers and animal manure of 175 Tg $year^{-1}$, animal manure N being 65 Tg $year^{-1}$ (37%) and N fertilizer 110 Tg $year^{-1}$. Hence, the total amount of animal manure in intensive systems has steadily increased between 1970 and 1995 and will continue to do so in the coming three decades, but its share in total inputs has decreased from 60% to 40% in the past three decades, and decrease at a slower rate in the coming three decades. The quantity of animal manure N available for spreading in intensive agricultural systems increased from 19 to 24 Tg $year^{-1}$ in the 1970–1995 period and according to the projection used it will increase further to 28 Tg $year^{-1}$ in 2030.

The current N input from biological N fixation varies from one region to another, with high values in North America and Latin America, Sub-Saharan Africa and Southeast Asia, where soybean and other leguminous crops are more important than in other regions. Inputs from N fixation have strongly

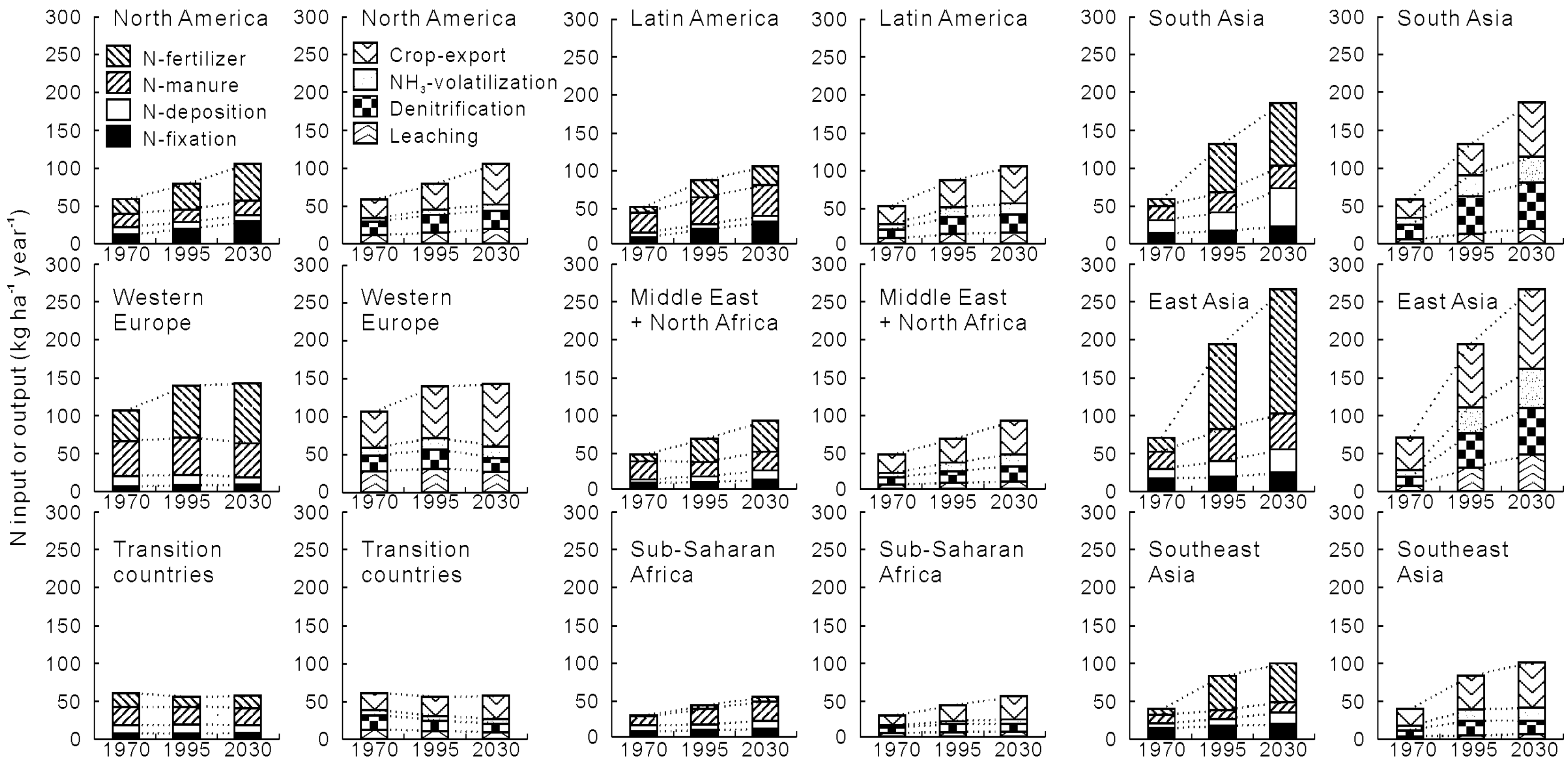

Fig. 4 N input (left hand) and output terms (right hand) of the surface N balance for nine world regions for 1970, 1995 and 2030 aggregated from 0.5 by 0.5 degree resolution grid maps. For each region the left-hand figure indicates the N inputs, and the right-hand figure the N output terms. See Fig. 3 for an explanation of the different balance terms.

increased between 1970 and 1995, and will continue to grow in all world regions in the coming three decades. This development is primarily related to the increasing demand for soybeans as an animal feed resource. The regions with most prominent growth in soybean production are North and South America and Sub-Saharan Africa (Table II and Fig. 4).

The estimated overall system N recovery for 1970 in a number of regions such as East Asia (61%), Latin America (51%) and Middle East and North Africa (54%) is high compared to the average for the industrialized countries of 44% (Table V). According to our results there has been a steady decrease of the overall system N recovery in the developing countries between 1970 (52%) and 1995 (42%), with 1995 values ranging from 31% (South Asia) to 55% (Southeast Asia). In industrialized countries the overall system N recovery slowly increased from 44% to 45% (Table III). Further increases are projected for 2030 to 45% in developing regions in the period 1995–2030, while in industrialized countries the efficiency will improve to 52% (Table V).

TABLE V

System N recovery and nitrogen use efficiency (NUE) for world regions for 1970, 1995 and 2030

Region	System N recovery[a]			NUE[b]		
	1970	1995	2030	1970	1995	2030
	%					
North America	42	43	51	49	48	63
Western Europe	44	49	58	44	54	68
Transition countries	38	46	55	43	67	83
Latin America	51	43	49	82	49	66
Middle East + North Africa	54	47	50	85	58	63
Sub-Saharan Africa	48	51	58	139	108	131
South Asia	42	31	38	99	41	58
East Asia	61	43	39	108	48	42
Southeast Asia	58	55	61	135	78	90
World	46	43	47	67	52	61
Developing	52	42	45	103	51	58
Industrialized	44	45	52	48	49	62

[a] System N recovery is calculated as the N in all harvested crops and grass cut and consumed by animals expressed as % of total inputs from N deposition, N fixation, N fertilizer and animal manure.
[b] NUE is calculated as the N recovery in upland crops and wetland rice (excluding leguminous crops) as % of N input from fertilizers and manure.

Regarding the NUE for upland crops and wetland rice there are also important differences between regions. The industrialized countries show a slow increase of NUE from close to 48% in 1970 to 49% in 1995. In the coming three decades NUE values will increase further, according to the projection used, to values of the order of 62% in the industrialized countries. In 1970 the transition countries showed NUE values of 43%, which is similar to those in Western Europe. However, after 1990 fertilizer use strongly decreased, and the NUE increased to 67%. For 2030 we project an increase of NUE in the transition countries to about 80%.

In the developing countries our results for 1970 indicate high values of NUE. For example, in Sub-Saharan Africa, Southeast and East Asia the NUE for 1970 exceeds 100%, while in other developing regions the NUE exceeds 80%. Between 1970 and 1995 the NUE values for developing countries have decreased. Whereas in Sub-Saharan Africa the NUE for 1995 still exceeds 100% and in Southeast Asia it is close to 80%, in other developing regions it sharply dropped between 1970 and 1995 to values comparable to those in industrialized countries in 1970.

For the coming three decades the average NUE value for the developing countries as a whole is projected to increase to 58%. However, there are large differences between regions and countries. The NUE for 2030 for Sub-Saharan Africa will exceed 100%, while the NUE for Southeast Asia will be 90%, which is comparable to the 1970 values of many other developing regions. The NUE for South Asia, Middle East and North Africa and Latin America will increase to the 1995 values for industrialized countries of 50%–60% or higher.

Some of the differences in the overall system N recovery and NUE can be explained from the characteristics of the crop production system. For example, low overall system N recovery and NUE in many Asian countries are related to paddy rice cultivation. Low recoveries in wetland rice are often due to difficulties in controlling N losses by ammonia volatilization and denitrification (Singh *et al.*, 1995).

Further differences between regional overall system N recovery and NUE are caused by the mix of crops. Regions such as Southeast Asia with a large proportion of crops with inherently low N content, such as sugar crops, root and tuber crops, vegetables and fruits (Table III), will show lower values for the system N recovery and NUE than regions with a large share of cereals, leguminous crops and oil crops with high N, such as the USA.

However, such differences are only apparent in the system N recovery and NUE when the overall intensity of the production system is similar. This is clearly not the case. Comparison of the levels and development of system N recovery and NUE shows that there are N inputs in many developing countries that we did not account for in our approach. One form of N inputs that we ignored is the use of night soil. For example, FAO (1977) estimated that in the 1970s the contribution of night soil in China was about 0.7 Tg N year^{-1}. This is about 20% of the amount of animal manure in intensive agricultural systems according to our estimate. Hence, in 1970 this agricultural N source was very important. If the amount of N from night soil has not changed, it would be about 8% of the amount of animal manure in 1995 and 2% of total inputs. Assuming that with continuing economic growth and increasing welfare the use of night soil is on the decrease, the underestimation of the inputs is less than 2%. The high values of the overall system N recovery and NUE for East Asia for 1970 indicate an underestimation of the N input. However, including the estimate for night soil would reduce the overall system N recovery to 58% in 1970. The system N recovery and NUE values for East Asia for 1995 are comparable with the value for North America, and may be more realistic. In other world regions, such as South Asia and Africa, the use of human waste is not a common practice.

A second unaccounted for N input may be the depletion of the soil N pool. In systems with low inputs and crop yields, soil organic matter may be lost by decomposition, and mineralization of the N may serve as an N source for plants. Nutrient depletion has been observed in many studies for Africa (Nandwa and Bekunda, 1998; Smaling *et al.*, 1993; Stoorvogel and Smaling, 1990; Stoorvogel *et al.*, 1993), South Asia (Singh *et al.*, 1995), East Asia (Sheldrick *et al.*, 2002) and South America (Koning *et al.*, 1997).

In Asia and Africa soil N depletion is probably the main cause for the high overall system N recovery and NUE values for 1970 (and for Southeast Asia and Sub-Saharan Africa for 1995 and 2030). The low N input systems in these world regions have been able to sustain a low total production volume at low levels of crop yields (FAO, 2001) but at the cost of soil fertility. In fact, crop yields have not increased substantially in many, primarily African, countries (FAO, 2001). It is not clear how important the role of shifting cultivation in maintaining the soil productivity has been in these regions, and how much forested land has been converted to agriculture to compensate for the productivity decline. Comparison with the results for industrialized countries for 1970–1995 indicates that values for NUE exceeding 70%–80% may be related to depletion of the soil N pools.

We recognize that the estimation of crop N export is fraught with potential errors, particularly the role of fodder crops, crop residues, animal manure and night soil: i) we ignored the production of fodder crops in non-OECD countries due to lack of data; ii) apart from the slight underestimation caused by this assumption, there may be other contributions to removal of N from agricultural fields which are not accounted for. These include stubble grazing and other uses of crop residues (*e.g.*, animal feed and fuel). For example, Sheldrick *et al.* (2002) assumed that 60% of crop residues is returned to the soil, and 40% is exported from the field, but recognized that no data is available to support this; iii) animal manure is increasingly generated in landless and intensive mixed systems, and it is uncertain how much is actually applied to crops and grassland. If in reality much of the collected and stored manure is not applied (*e.g.*, many lagoon storage systems in the USA), we overestimate the animal manure N input and underestimate the overall system N recovery. This may also be the case in South and East Asia in 2030. It should be noted that animal manure that is not returned to the agricultural cycle can also cause environmental problems, for example by ammonia volatilization and leaking to groundwater from storage systems such as lagoons and other reservoirs; and, iv) ignoring the use of night soil (as discussed above) causes an overestimation of the overall system N recovery in particularly in 1970 in East Asia and possibly other (Asian) regions as well. Decreasing use of human waste as fertilizer generally implies an increasing input of N (and phosphorous) to surface water, depending on the degree and type of sanitation and wastewater treatment.

The surface N balance aggregated for all crops indicates strong variability between countries and also within countries (Fig. 5). The surplus of the surface N balance is the largest in countries with multiple cropping (*e.g.*, China, India) and in countries with highly intensive agricultural production (The Netherlands, Belgium, and Denmark). Countries with low surpluses (< 10 kg N ha^{-1} year^{-1}) and N deficits are located in Latin America, Africa and Asia.

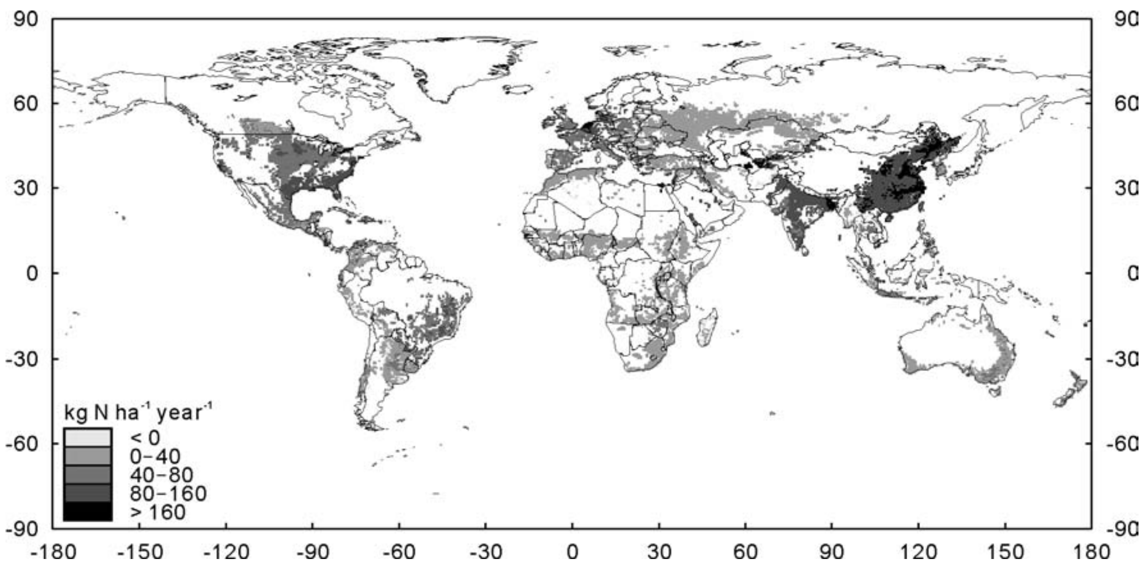

Fig. 5 Surface N balance for 1995 aggregated from the 0.5 by 0.5 degree data for upland crops, wetland rice and leguminous crops.

Ammonia volatilization from fertilizer and manure application and grazing (excluding NH_3 loss from animal housing, feedlots, other storage and collection systems) ranges from values of about 8%–9% of total inputs in temperate regions to 12%–21% in tropical regions. Differences are related to climate, type of crop, mix of fertilizers, and the importance of animal manure spreading. Firstly, ammonia volatilization rates are higher under warm than under cold conditions. Secondly, the importance of wetland rice cultivation and the widespread use of urea in Asia lead to high NH_3 volatilization losses of $> 20\%$ of the fertilizer N inputs. Thirdly, NH_3 volatilization associated with animal manure application is of the order of 20%–30% of the N applied. In most developing regions the livestock production and associated manure will increase in the coming decades, particularly in landless and mixed systems. This implies a fast growth of availability of manure for application (Fig. 4) and associated NH_3 volatilization.

The variation of leaching rates (roughly 5%–20% of total N inputs) is smaller for the aggregated world regions than for individual countries. Mean regional leaching losses for 1995 vary from low values in dry climates (*e.g.*, South Asia, 10% of total N inputs) to close to 20% in more humid, temperate climates (*e.g.*, Western Europe, 22%). World regions dominated by warm and wet climates (Asian regions) also show relatively low leaching rates ($< 16\%$ of total N inputs) due to high denitrification losses (24% in East Asia). There are decreasing trends in both denitrification and leaching losses between 1970 and 2030 in North America and Western Europe as a result of growing recovery rates, and in all world regions between 1995 and 2030.

CONCLUSIONS

While the N recovery hardly increased in the industrialized countries between 1970 (44%) and 1995 (45%), the values for developing countries started from 52% in 1970 and decreased to 42% in 1995. For the coming three decades our results indicate a further increase to 52% in the industrialized countries and an increase to 45% in developing countries. The nitrogen use efficiency (NUE) for industrialized countries increased from 48% in 1970 to 49% in 1995 and is projected to increase to 62% in 2030. In the developing countries there has been a decrease of NUE from 103% in 1970 to 51% in 1995. The projected NUE for 2030 for developing countries is 58%, with large differences between individual countries.

In many developing countries there is an N balance deficit, causing high overall system N recovery. Many countries have NUE values exceeding 100%, mainly in Sub-Saharan Africa and Southeast Asia. This is related to declining soil N pools, and thus to declining soil fertility. In the coming three decades the N balance deficit will gradually decrease in most developing regions, except for Sub-Saharan Africa ($>$ 100%) and Southeast Asia (90%). Environmental losses of N are smaller for systems with an N deficit than for situations with surplus. However, the loss of soil productivity by depletion of the soil N (and P) pools is a severe environmental problem in itself, and it may also cause forest clearing and conversion to agriculture to compensate for production loss.

Our results suggest that for countries with large surpluses additional gains in the system N recovery and nitrogen use efficiency could potentially be achieved in intensive production systems in developing countries by improved management practices. However, the N recovery and NUE depend on climatic conditions (precipitation, temperature) that influence crop growth and nutrient uptake, ammonia volatilization, denitrification and leaching rates. The potential efficiency is therefore not the same for all countries. Moreover, efficiencies are also determined by the mix of crops. Some crops have low N contents and thus inherently low NUE values. For other crops such as wetland rice it is difficult to achieve high NUE due to the specific conditions in inundated rice fields.

Apart from the overall increase in fertilizer use and livestock production, there is also a global trend towards concentration of agricultural activities related to livestock and intensive crop production and horticulture in peri-urban areas (Bouwman, 1997). This will lead to large local surpluses of N and P from animal manure and associated losses to aquatic systems and atmosphere. Hence, this will turn nonpoint source pollution into a more local problem.

Despite the improvement of the efficiency, it is clear that the fast increase of production and intensification will almost inevitably lead to increasing environmental losses of N. Depending on the characteristics of the system, there will be increasing emissions of NH_3 to air, nitrate leaching to groundwater, and release of gaseous N oxides produced during denitrification which play an important role in greenhouse warming (N_2O) and ozone chemistry (NO).

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