A comparison of global spatial distributions of nitrogen inputs for nonpoint sources and effects on river nitrogen export

G. Van Drecht,[1] A. F. Bouwman,[1] E. W. Boyer,[2] P. Green,[3] and S. Siebert[4]

Received 7 January 2005; revised 26 April 2005; accepted 24 May 2005; published 30 July 2005.

[1] We compared four global data sets for the year 1995 for nonpoint N sources with 0.5° by 0.5° spatial resolution. Data were developed to drive models for assessing the river export of nitrogen (N) at the global scale. The data include annual N inputs (biological N fixation, atmospheric N deposition, N fertilizer, animal manure, and human emissions) and outputs (ammonia volatilization and N removed from agricultural fields by harvesting crops and grass consumption). There are important differences at the global, regional, country, and river-basin scales for all input and output terms in the data sets. The main differences are in the rates and the spatial allocation for biological N fixation and atmospheric N deposition, animal manure inputs and management, and N in harvested crops and grass consumption. Inputs of N fertilizer in agricultural systems are relatively well known at the country scale (and subnational scale for some large countries), but their spatial allocation also shows major differences between the four data sets. The level of disagreement between the different data sets increases with decreasing river basin size, which is related to the difficulty of spatial allocation when river basins cover only a few grid cells. Transport efficiencies to calculate river N export from the N surplus obtained from a regression approach and from a meta model derived from a conceptual model are in good agreement when aggregated to continents and receiving oceans.

Citation: Van Drecht, G., A. F. Bouwman, E. W. Boyer, P. Green, and S. Siebert (2005), A comparison of global spatial distributions of nitrogen inputs for nonpoint sources and effects on river nitrogen export, *Global Biogeochem. Cycles*, *19*, GB4S06, doi:10.1029/2005GB002454.

1. Introduction

[2] Global river nitrogen (N) export to coastal oceans has doubled in the twentieth century [*Vitousek et al.*, 1997]. This has caused eutrophication of many coastal and estuarine ecosystems leading to chronic hypoxia, reductions in species diversity, and stressed fisheries resources [*Vollenweider et al.*, 1992]. The large increases of N inputs to surface water related to food production and human waste that are expected to occur globally in the twenty-first century [*Galloway et al.*, 2004] have induced many research groups to investigate N transport in large watersheds and export to coastal marine ecosystems [*Howarth et al.*, 1996; *Seitzinger and Kroeze*, 1998; *Garnier et al.*, 2001; *Boyer et al.*, 2002; *Garnier et al.*, 2002; *Seitzinger et al.*, 2002; *Galloway et al.*, 2004; *Green et al.*, 2004; *Bouwman et al.*, 2005a].

[3] Common in all these approaches is the development of models to describe the N inputs from nonpoint and point sources and to estimate the fate of the N in the hydrological systems up to the river mouth. Point sources are primarily associated with sewage effluents, including human emissions and wastewater from industrial activities, and generally located in urban areas. Nonpoint sources comprise all N entering the surface water system in a diffuse manner and are associated with agricultural land use, households, and (semi-)natural ecosystems.

[4] Most global models concentrate on the mean annual riverine export from large river basins to the coastal sea. Some regional-scale models have been compared by *Alexander et al.* [2002]. Our study was part of an international interdisciplinary effort to model river export of multiple bioactive elements (C, N, P, and Si) and elemental forms (dissolved/particulate, inorganic/organic) called Global Nutrient Export from Watersheds (Global NEWS). We focus on the global-scale N input data and models that have been developed in the framework of this project.

[5] Specifically, we present and analyze the inputs and output terms used for estimating the N surface balance for the base year 1995, and their spatial distribution for the nonpoint sources of N, comparing global data sets devel-

[1]Netherlands Environmental Assessment Agency, National Institute for Public Health and the Environment, Bilthoven, Netherlands.

[2]Department of Environmental Science, Policy, and Management, University of California, Berkeley, California, USA.

[3]Complex Systems Research Center, Institute for the Study of Earth, Oceans, and Space, University of New Hampshire, Durham, New Hampshire, USA.

[4]Institute of Physical Geography, University of Frankfurt, Frankfurt, Germany.

0886-6236/05/2005GB002454$12.00

oped by *Bouwman et al.* [2005a], *Boyer et al.* [2004], *Green et al.* [2004], and *Siebert* [2005]. Data provided by *Bouwman et al.* [2005a], *Green et al.* [2004], and *Siebert* [2005] allow for making up surface balances for the nonpoint sources of N, while *Boyer et al.* [2004] estimate the amount of "new" N that is fixed and added to the global terrestrial system by anthropogenic activities. The data used in this paper have a spatial resolution of 0.5° by 0.5°.

[6] The input terms in the surface balance for the nonpoint sources are N fertilizer, animal manure N, biological N fixation, and atmospheric N deposition. The estimation and spatial allocation of each of these categories involves specific problems and uncertainties.

[7] The data on N fertilizer use are much more certain than those for the other N inputs. However, the spatial allocation of fertilizer use in agricultural production systems at the subnational scale is much more difficult than obtaining estimates at the country, regional, or global scale [*Bouwman et al.*, 1999]. An additional problem is that in many parts of the world, primarily in developing countries, a considerable part of the cultivated area does not receive inputs from N fertilizers [*Bouwman et al.*, 2002b] and a minor part is intensively cultivated with high fertilizer inputs.

[8] The inputs of animal manure are less certain than N fertilizer use, because the estimation of the production of animal manure and its use involves several steps, each adding more uncertainty. The manure production is estimated from the animal populations and N excretion rates by animal category, management of the manure in different systems, and associated ammonia (NH_3) volatilization in each system. The spatial allocation of animal manure in the various management systems is an additional, major cause of uncertainty [*Bouwman et al.*, 1999].

[9] For the other input terms, both the quantification and the allocation are fraught with uncertainties. N fixation rates in both natural and agricultural systems are difficult to quantify, their controls are poorly known, and they vary strongly due to heterogeneity of the environmental and management conditions [*Vitousek et al.*, 2002]. Also, N fixation rates may change in time due to succession, and fixation rates may be much lower in mature forests than in growing secondary forests [*Vitousek et al.*, 1988, 1989].

[10] Atmospheric N deposition rates are generally obtained from coarse-resolution chemistry-transport models (CTM) that are driven by emission inventories that may not be consistent with the land cover and land use data used to compile the data sets for which the deposition fields were made. In addition, it is difficult to generate estimates for short-range deposition. Deposition velocities strongly depend on the surface roughness, which varies strongly within grids of these CTMs. There is a lack of measurements of deposition rates in many parts of the world, and finally, the validation of coarse-scale models with point measurements involves serious scaling problems [*Bouwman et al.*, 1997].

[11] The output terms of the surface balance used by the different data sets include NH_3 volatilization and export of N from the field in the form of harvested products and grazing. The uncertainties of estimates for NH_3 volatilization are discussed in detail by *Bouwman et al.* [1997, 2002a]. Crop production data by country are readily available from statistics. However, the main uncertainty in estimating crop export is the N content of crops, which depends on a host of factors including the crop variety used, climate, soil fertility, crop yields, and management. All these factors are strongly variable under field conditions. Further uncertainties are related to the use of crop residues (incorporation, field burning, fuel, animal feed, etc.), for which no statistics are available. For the category of fodder crops, there are no production data available, and we do not have reliable global data on grazing by ruminants, and their grass and N uptake.

[12] Given all these uncertainties, it is clear that it is not possible to select the "best" set of data. Moreover, the different data sets rely on the same data sources for most of the input and output terms, the major difference being the spatial allocation. Therefore, by comparing the data, we may identify the strengths and weaknesses of different approaches and underlying data.

[13] Since the comparison involves many N input and output terms of the surface balance, and because we also analyze effects of differences in the spatial allocation of the different N balance terms, it is not possible to do a sensitivity analysis. Instead, we will first compare the differences in the estimates of the surface balance and individual N input and output terms, and their spatial allocation on the global, regional, country, and river-basin scale. The latter scale is the focus of most participants of the NEWS project [e.g., *Green et al.*, 2004], while other models require the grid-scale for calculating the fate of the N in the hydrological system [*Van Drecht et al.*, 2003].

[14] We will also investigate the effect of model differences. We use a meta version of the model developed by *Van Drecht et al.* [2003] and compare this model in terms of transport efficiencies with those of *Green et al.* [2004] at the global and continental scale. Finally, river N export will be estimated using this meta model to investigate the effects of differences in the surface balance N surplus and their spatial allocation.

2. Data and Methods

2.1. Data Sets Used

[15] In this paper we compare the input data sets for the nonpoint sources developed by *Bouwman et al.* [2005a] (the data set implemented in the river N export model of *Van Drecht et al.* [2003], referred to as BOUW), *Boyer et al.* [2004] (BOYE), *Green et al.* [2004] (GREE), and *Siebert* [2005] (SIEB) (Table 1). These data sets can be obtained from the authors.

[16] The SIEB, BOYE, and BOUW data are part of modeling approaches designed for scenario analysis. The GREE data have been used to analyze historical (pre-industrial) and contemporary river N export.

[17] The spatial resolution of the data used for this paper is 0.5° by 0.5°, with a temporal resolution of 1 year, for this comparison the year 1995. The N inputs considered in BOUW, GREE, and SIEB include biological N fixation (N_{fix}), atmospheric N deposition (N_{dep}), application of synthetic N fertilizer (N_{fert}), animal manure application

Table 1. Approaches and Data Used in the BOUW, BOYE, GREE, and SIEB Data Sets for Land Cover, Cropping Intensity, N Fertilizer Use, Animal Manure Production and Management, Biological N Fixation in Agricultural and Natural Ecosystems, Atmospheric N Deposition, N Export From Agricultural Fields, Ammonia Volatilization and Net Trade of N in Agricultural Products

Data Type	BOUW	BOYE	GREE	SIEB
Land cover	0.5° resolution [*IMAGE-team*, 2001; *Bouwman et al.*, 2005b] for arable land, grassland and natural ecosystems; a grid cell is covered by urban area and either agriculture (with distribution of seven crop groups and grassland) or natural ecosystems	1-km resolution land cover database including cropland [*Land Processes Distributed Active Archive Center*, 2003].	1-km resolution land cover database including cropland and grassland [*Resource Observation Systems* (*EROS*) *Data Center*, 2000]; 0.5° resolution data for undisturbed vegetation [*Melillo et al.*, 1993]; 1-km resolution city lights data set [*Elvidge et al.*, 1997] defining urban areas	0.5° resolution from combination of *IMAGE-team* [2001], *United States Geological Survey* (*USGS*) [2000] and *Ramankutty and Foley* [1999]; each cell has distribution of irrigated and rainfed areas for seven crop groups, four biofuel crops, grass + fodder, other crops, pasture, natural ecosystems incl. managed forests
Cropping intensity	Country-level data [*FAO*, 2001].	country-level data [*FAO*, 2001], used in the net import of N in food & feed.	country-level data [*FAO*, 2001]	data for world-regions (17) from *IMAGE-team* [2001]
N fertilizer use	mean country N application rates for wetland rice, leguminous crops, other upland crops and grassland [*FAO*, 2001; *IFA*, 2002; *IFA/IFDC/FAO*, 2003]	country N fertilizer consumption totals [*FAO*, 2002] evenly distributed over cropland (see above) and resampled to 0.5° resolution.	country N fertilizer consumption totals [*FAO*, 2001] evenly distributed over cropland (see above) and resampled to 0.5° resolution	mean country N application rate per crop group [*IFA*, 2003; *IFA/IFDC/FAO*, 2003] accounting for fallow and unfertilized areas; country total fertilizer N use may not agree with total country N use [*FAO*, 2001]
Animal manure production	country animal populations (dairy and nondairy cattle, buffaloes, sheep, goats, pigs, poultry, horses, asses, mules and camels) [*FAO*, 2001] and regional species-specific N excretion rates [*Van der Hoek*, 2001]	not applicable; accounted for in the net import of N in food and feed	the 1° resolution map of domesticated animals [*Lerner et al.*, 1988] was updated with data for 1995 with data from [*FAO*, 2001] with species-specific N excretion rates based on animal intake [*National Research Council*, 1985, 1989, 1998, 2000a, 2000b; *Smil*, 1999]	country animal populations (dairy and non-dairy cattle, buffaloes, sheep, goats, pigs, camels, horses, chicken, turkey, ducks, geese) [*FAO*, 2003] on a 0.5° resolution[a]; N excretion rates for cattle per country based on cattle productivity and constant rates for the other animal categories from *Bouwman et al.* [1997] and *Brandjes et al.* [1996].
Animal manure management	country estimates for grazing (based on feed intake), manure storage, and other uses (fuel, unused manure) [*Bouwman et al.*, 2005c]; ammonia loss from stored manure is excluded from the manure N available for spreading	not applicable; accounted for in the net import of N in food and feed	not applicable; accounted for in the crop production N content	manure storage depends on animal category, world regional (17) preferences for specific fodder types [*IMAGE-team*, 2001], and simulated fodder availability in pastures; stored manure is applied to cropland and managed pasture
Biological N fixation in agriculture	country-level legume-crop production combined with N content in harvested parts for pulses and soybeans (global estimates) [*Bouwman et al.*, 2005c], and global mean N fixation rates for grassland and cropland (5 kg N ha^{-1}) and wetland rice (25 kg N ha^{-1}) from *Smil* [1999]	country-level legume-crop production total from *FAO* [2002], weighted by global mean fixation rates reported by *Smil* [1999]	country level legume-crop area [*FAO*, 2001] and global mean N fixation rates [*Smil*, 1999] evenly distributed over cropland and aggregated from 1 km to 0.5° resolution	for pulses, soybeans and clover based on soil N balance (high fixation rates in case of N-deficit, low for N-surplus), and global mean N fixation rates for grassland and cropland (5 kg N ha^{-1}) and wetland rice (25 kg N ha^{-1}) [*Smil*, 1999]

Table 1. (continued)

Data Type	BOUW	BOYE	GREE	SIEB
Biological N fixation in natural ecosystems	ecosystem-specific N fixation rates based on the inventory of *Cleveland et al.* [1999]	computed by C. Cleveland and G. Asner (unpublished data, personal communication, 2002) for undisturbed vegetation classes; estimates are based on an updated inventory following *Cleveland et al.* [1999], coupled with the TerraFlux biogeochemical model of *Asner et al.* [2001], which constrains fixation estimates by the amount of nitrogen required by plants to support net primary production	potential N fixation rates defined for undisturbed vegetation classes from *Cleveland et al.* [1999] and A. Townsend (personal communication, 2001)	N fixation rates computed as function of actual evapotranspiration and scaled for each vegetation type to match with the central estimate of *Cleveland et al.* [1999]
Atmospheric N deposition	long-range N transport and deposition from the STOCHEM global chemistry-transport model [*Collins et al.*, 1997] with 5° by 5° resolution, converted to 1° and smoothed, and includes short-range dry deposition [*Bouwman et al.*, 2002c]	long-range transport of wet and dry atmospheric NO_y and NH_x deposition from the TM3 global chemistry-transport model for 5° by 3.75° resolution [*Dentener and Crutzen*, 1994]	simulated total (wet + dry) NO_y plus NH_3 modeled deposition for 1990 from MOGUNTIA [*Dentener and Crutzen*, 1994]; original data sets were provided at 1° resolution smoothed from the coarser original MOGUNTIA 7.5° (lat) × 10° (lon) resolution; data were further resampled to 1 km and 0.5° resolution	as in BOYE, long-range transport of wet and dry atmospheric NO_y and NH_x deposition is from the TM3 global chemistry-transport model for 5° by 3.75° resolution [*Dentener and Crutzen*, 1994]; fields were interpolated to 0.5° by 0.5° resolution; NH_x deposition modified to include NH_3-emissions (manure, fertilizer) assuming 30% short-range deposition and 70% long-range transport
Nonpoint human N emissions	N from human waste from inhabitants not connected to sewerage systems was assumed not to end in surface water	not applicable; accounted for in the net import of N in food and feed	human N excretion derived from protein intake per capita [*FAO*, 2001], the fraction urban population connected to sewerage systems [*World Resources Institute*, 1998; *United Nations Settlements Programme*, 2001], and the level of wastewater treatment for sewered populations [*Organization for Economic Co-Operation and Development*, 1999]; rural and nonsewered urban populations were treated as diffuse sources, distributed over urban and rural population using *Vörösmarty et al.* [2000]	not considered
N export from arable fields	country-level crop production data [*FAO*, 2001] and N content for 34 crops used by *Bruinsma* [2003] aggregated to the broad categories wetland rice, leguminous and other upland crops; estimates include fodder crops	not applicable; accounted for in the net import of N in food and feed	country-level crop production data [*FAO*, 2001] for seven groups (cereals, legumes, sugar crops, roots and tubers, vegetables and fruits, forages, and other crops) and N contents [*Smil*, 1999]; crop N was distributed over cropland (1 km), weighted by atmospheric inputs and aggregated to 0.5° resolution	simulated crop yields for seven crop groups scaled to production data for 17 world regions [*IMAGE-team*, 2001] on a 0.5° by 0.5° grid, combined with N content of the harvested parts; crop residue management for world regions according to *IMAGE-team* [2001]

Table 1. (continued)

Data Type	BOUW	BOYE	GREE	SIEB
N export from grassland	calculated as 0.6 times the sum of the N inputs minus the NH_3 emission	not applicable; accounted for in the net import of N in food and feed.	accounted for as local grazing animal intake requirements not exceeding N content in grassland at 0.5° resolution	based on feed-dry matter (energy) demand per animal category [*IMAGE-team*, 2001], livestock density, and dry matter availability on grassland and pasture
Ammonia volatilization	volatilization from stored manure (not included in surface balance) and grazing according to *Bouwman et al.* [1997]; N fertilizer and manure application according to *Bouwman et al.* [2002a]	not applicable; used long-range modeled NH_x deposition only	total volatilization loss from animal waste from *Smil* [1999], and from fertilizers calculated for 1 km and aggregated to 0.5° according to *Bouwman et al.* [2002a]	volatilization loss from stored manure, grazing, manure spreading, and different fertilizer types according to *Bouwman et al.* [1997], country fertilizer mix from *FAO* [2003]
Net import (or export) of N in human food and animal feedstocks	not applicable	country-level imports and exports for each major crop and animal type [*FAO*, 2002] weighted by their N contents [*Lander and Moffitt*, 1996; *Bouwman and Booij*, 1998]	not applicable; accounted for in balance between crop/livestock production and human/animal consumption estimated by country [*Green et al.*, 2004]	not applicable

[a]From original 2.5-min resolution livestock densities [*Gerber*, 2004] and subnational and national livestock population data as described in detail by *Siebert* [2005].

and animal N excreted during grazing (N_{man}), and inputs of nonpoint human emissions to surface water (N_{humnp}). Outputs considered include N removal from the field by crop harvesting, cutting of hay and grass and grass consumption by grazing ruminants (N_{exp}), and NH_3 volatilization (N_{vol}).

[18] Our data comparison focuses on the N surplus of the surface balance (N_{sur}), since this quantity was used in different studies [*Van Drecht et al.*, 2003; *Green et al.*, 2004; *Bouwman et al.*, 2005a] as a basis for calculating riverine N export. This is because all the terms used in the surface balance are relatively well known compared to the processes of denitrification and transport in soils, groundwater, and surface water systems, and surface balance surpluses generally agree fairly well with estimates obtained from *Organization for Economic Co-operation and Development* (*OECD*) [2001]. N_{sur} (kg yr^{-1}) is calculated for each 0.5° by 0.5° grid cell as follows:

$$N_{sur} = N_{fix} + N_{dep} + N_{fert} + N_{man} + N_{humnp} - N_{exp} - N_{vol}. \quad (1)$$

Where there are important discrepancies we will also compare the individual input or output terms. The four approaches are summarized in Table 1. Some aspects of the data need to be discussed in more detail.

[19] In the BOYE data set the new reactive N inputs are considered, including N fertilizer use, N fixation, N deposition, and net trade of products. Net trade is considered to reflect a mass balance of needs versus production and inherently includes food production and waste. Because of this approach, the BOYE data set cannot be used for the comparison of the N surplus. Instead, we compare the spatial data on N fertilizer use, biological N fixation, and atmospheric N deposition for the different data sets.

[20] While BOUW and GREE use a static surface balance approach, based on the assumption that there is no change in the soil N pools, the SIEB data are generated by a dynamic model that describes soil N pool changes. This may cause problems in the comparison for grid cells where net soil N depletion or accumulation is calculated. In our comparison we assign zero values to grid cells with negative values for N_{sur}.

[21] While the BOUW data are based on the assumption that the N in waste from (rural) population not connected to sewerage systems does not end in surface water, the GREE data also include the nonpoint N inputs from human waste to surface water (N_{humnp}). This aspect is not considered in the SIEB data.

[22] The four data sets are based on different spatial distributions of land cover and land use. The SIEB, BOYE, and GREE land cover data sets were originally developed at a finer spatial resolution than the 0.5° by 0.5° used in this comparison (Table 1).

[23] In all the data sets the input data for atmospheric deposition are obtained from different atmospheric chemistry-transport models with a model resolution of 7.5° by 10° (GREE), 5° by 5° (BOUW), and 5° by 3.75° (BOYE and SIEB) (Table 1). BOUW and SIEB included short-range dry deposition of NH_x using inventories of NH_3 emissions. This subgrid dry deposition parameterization was done because the resolution of the STOCHEM, MOGUNTIA, and TM3 models used was considered too coarse to resolve this process. Two data sets (BOYE and SIEB) are based on the same model, but main differences are caused by short-range deposition (included in SIEB and ignored in BOYE) and the smoothing (not smoothed in BOYE and smoothed to 0.5° resolution by SIEB; see Table 1).

[24] The four data sets for biological N fixation rates were calculated with similar methods, albeit different characterizations of the spatial distribution of agricultural and natural vegetation classes over which to assign the rates of biological N fixation reported in the literature. For biological N fixation rates in cultivated crop lands, the data sets are based on agricultural fixation rates reported by *Smil* [1999] and others (see Table 1). For natural biological N fixation rates in noncultivated vegetated lands, the approaches are based heavily on a recent compilation of fixation rates observed in natural ecosystems reported by *Cleveland et al.* [1999]. BOUW uses biological N fixation rates by ecosystem directly. BOYE, GREE, and SIEB use data sets where the reported fixation rates were coupled with ecosystem process models to constrain the estimates of natural biological N fixation by plant biogeochemistry and the water balance.

[25] Application rates of fertilizer N are calculated from the total fertilizer use per country in 1995 and the area of cropland (GREE, BOYE), or by assigning data on fertilizer use by crop and allocating these values to the respective crop areas (BOUW, SIEB). None of the data sets provide N application rates specific for irrigated and rainfed areas. However, the SIEB data account for fallow and unfertilized cultivated land (Table 1).

[26] The data on manure N from livestock excreta used by GREE are taken from a 1° by 1° resolution data set developed by *Lerner et al.* [1988]. The three other data sets use finer grids (Table 1). Apart from the resolution used, there are also differences related to production systems. GREE does not distinguish between different livestock production systems within countries. BOUW is based on pastoral grazing and mixed/industrial livestock production systems with differentiation of animal manure management at the country scale. SIEB uses 1 by 1 km spatial livestock distributions which are scaled using country and subnational data on livestock numbers, with no specification of the animal manure management. The N excretion rates used in the GREE data set are generally lower than those in BOUW and SIEB. Particularly in countries with intensive and industrial livestock production, the GREE data may underestimate N excretion rates [*Van der Hoek*, 1998, 2001].

[27] BOUW did not include NH_3 emissions from stored manure in their surface N balance approach, arguing that this is a point source which was excluded from the manure N available for spreading. SIEB and GREE present the NH_3 volatilization for the livestock production as a whole, and do not deduce the N-loss from manure storage. Hence we can safely compare the difference between total manure N and NH_3 volatilization (equation (1)) for all the data sets.

[28] The GREE data set is the only one that explicitly represents data for human nonpoint sources of N (i.e., human waste effluents that enter surface water in a diffuse manner, not via sewerage systems) (Table 1). This N source is assumed not to end in surface water in the BOUW data,

Table 2. Land Area in the Common Land Mask, Sum of N_{sur} (*tm*) for BOUW, GREE, and SIEB, and the Fraction of Total Mass of N_{sur} With Match (*m*), and Pearson Correlation Coefficient (*r*) for the Three One-by-One Comparisons of the Data Sets for the World, World Regions, and Selected Countries

Region/Country	Area, 10^6 Km^2	BOUW *tm*, Tg yr^{-1}	GREE *tm*, Tg yr^{-1}	SIEB *tm*, Tg yr^{-1}	BOUW/GREE *m*	BOUW/GREE *r*	BOUW/SIEB *m*	BOUW/SIEB *r*	GREE/SIEB *m*	GREE/SIEB *r*
World	128.1	232.2	210.8	252.8	0.74	0.73	0.75	0.74	0.73	0.75
Canada	9.0	7.6	5.2	4.9	0.75	0.81	0.74	0.65	0.70	0.62
United States	9.1	20.4	16.9	19.4	0.78	0.78	0.80	0.75	0.77	0.69
Central America	2.6	5.6	4.9	8.2	0.76	0.24	0.73	0.39	0.68	0.20
South America	17.4	40.0	25.3	44.7	0.69	0.51	0.80	0.52	0.69	0.54
North Africa	5.7	5.5	4.9	2.9	0.76	0.77	0.54	0.84	0.65	0.87
Western Africa	11.2	22.0	17.2	24.9	0.69	0.41	0.78	0.76	0.64	0.55
Eastern Africa	5.8	10.1	7.8	14.1	0.70	0.25	0.74	0.63	0.63	0.26
Southern Africa	6.7	10.6	10.8	15.0	0.65	0.27	0.71	0.56	0.72	0.26
Western Europe	3.5	12.2	12.7	13.3	0.81	0.79	0.67	0.53	0.74	0.69
Eastern Europe	1.1	3.7	4.4	2.5	0.85	0.36	0.73	0.06	0.66	−0.14
Former USSR	21.6	20.1	16.8	15.1	0.76	0.72	0.74	0.69	0.73	0.68
Middle East	5.8	6.5	6.6	6.0	0.79	0.50	0.74	0.40	0.77	0.52
South Asia	5.0	21.8	24.2	27.3	0.88	0.87	0.85	0.83	0.84	0.76
East Asia	11.0	24.3	33.4	29.5	0.70	0.64	0.70	0.62	0.78	0.80
Southeast Asia	4.1	12.2	9.6	11.5	0.77	0.25	0.79	0.47	0.70	0.13
Oceania	8.1	8.5	9.3	12.4	0.64	0.24	0.64	0.40	0.62	0.31
Japan	0.3	1.1	0.9	1.3	0.81	0.33	0.80	0.49	0.75	0.39
Australia	7.5	6.9	7.8	10.0	0.63	0.06	0.62	0.37	0.62	0.22
Brazil	8.4	24.1	13.2	24.2	0.68	0.05	0.82	0.08	0.70	0.20
India	3.2	16.3	17.9	20.5	0.90	0.80	0.85	0.67	0.85	0.60
China	9.2	23.3	31.2	26.9	0.70	0.61	0.71	0.61	0.79	0.79

and it was not considered by SIEB. The approach used by BOYE considers the net input of N from food imports, which implicitly includes the recycling of both point and nonpoint sources of human waste.

[29] We used the land mask with the minimum coverage of reported N_{sur}, which is the GREE data set providing data for 58,819 0.5° by 0.5° grid cells. This is to avoid comparison of grid cells where data are not provided in one data set, while there is coverage in the others, or where it is not clear what is indicated by zero values (no data, or zero). This leads to an underestimation of the BOUW, BOYE, and SIEB inputs on the global scale, although the difference with the full coverage data sets is very small (see differences in *tm* for each data set in Tables 2 and 3).

2.2. Comparison of N Inputs and Outputs

[30] We consider different scales, including the global, regional, country, and river basin scales. For each data set and each scale, we computed the total mass *tm* (kg yr^{-1}) which is the sum of N_{sur} for all 0.5° grid cells within the area considered,

$$tm = \sum N_{sur}. \tag{2}$$

For each one-to-one (x, y) comparison, we calculated the Pearson correlation coefficient (r),

$$r = \frac{\sum(x-\bar{x})(y-\bar{y})}{\sqrt{\sum(x-\bar{x})^2 \sum(y-\bar{y})^2}}, \tag{3}$$

where $\bar{x}$, $\bar{y}$ represent average of x and y, and the mass m is the matching-mass fraction for N_{sur} in two data sets calculated according to *Janssen and Heuberger* [1995] as follows:

$$m = \frac{2\sum \min(x,y)}{\sum(x+y)}. \tag{4}$$

The fraction m theoretically ranges between 0 and 1. However, m is 0.4–0.5 when one data set in a one-to-one comparison is randomized. In our comparison, m typically ranges between 0.5 and 0.8, where m = 0.5 indicates absence of any agreement.

[31] The Pearson correlation coefficient (r) provides information on the differences between maps at the grid scale, whereby large differences between maps are reflected much more than in m. A disadvantage of r is that it yields low values when two data sets are clustered around a small range in both x and y direction. In that case, m may be high, indicating a good agreement.

2.3. Comparison of River N Export Models

[32] Two river N export models are compared, i.e., the global model developed by *Van Drecht et al.* [2003] and the model developed by *Green et al.* [2004]. The *Van Drecht et al.* [2003] model uses data on climate, soil properties, and hydrological characteristics of the groundwater system to describe NH_3 emissions, soil denitrification and leaching of nitrate from the root zone, N transport and denitrification in groundwater, and N retention in river systems. The uncertainty of the model results and the sensitivity to variation in the input data is discussed by *Van Drecht et al.* [2003]. The *Green et al.* [2004] model is a nonlinear regression model which takes into account N delivery by soils, groundwater, rivers, lakes, and reservoirs.

Table 3. Total N Inputs From Atmospheric N Deposition (N_{dep}), Biological N Fixation (N_{fix}), N Fertilizer (N_{fert}), and Animal Manure (N_{man}), and Outputs (Crop and Grass Harvest and Grazing, N_{exp}, and Ammonia Volatilization, N_{vol}) and Total Mass (*tm*) for Selected World Regions[a]

Source	Region	N_{dep}	N_{fix}	N_{fert}	N_{man}	N_{humnp}	Total N Input	N_{vol}	N_{exp}	Total N Output	tm[b]
BOUW	World	80.8	138.0	82.9	81.5	0.0	383.2	31.6	115.1	146.7	236.5
GREE	World	43.4	110.2	78.3	81.5	15.3	328.7	35.8	79.4	115.2	213.5
SIEB	World	68.6	160.1	72.3	107.7	...	408.7	40.4	109.0	149.4	260.8
BOYE	World	60.3	133.4	81.1	...	...	...	...	...	...	...
BOUW	Canada	2.2	5.1	2.0	0.9	0.0	10.2	0.4	1.9	2.3	7.9
GREE	Canada	1.8	2.5	1.6	0.8	0.0	6.8	0.4	1.1	1.5	5.3
SIEB	Canada	1.1	4.1	1.3	0.9	...	7.4	0.4	1.8	2.3	5.2
BOYE	Canada	1.2	7.3	1.6	...	...	...	...	...	...	...
BOUW	South America	9.5	30.0	3.1	14.5	0.0	57.1	3.3	13.6	16.9	40.2
GREE	South America	4.8	20.7	2.3	10.2	0.5	38.6	3.8	9.3	13.1	25.5
SIEB	South America	7.7	34.5	2.6	15.5	...	60.3	4.6	10.6	15.2	45.2
BOYE	South America	6.9	25.9	3.8	...	...	...	...	...	...	...
BOUW	North Africa	1.2	4.3	0.9	1.1	0.0	7.5	0.5	1.5	2.0	5.6
GREE	North Africa	0.7	3.0	1.2	0.8	0.3	6.1	0.4	0.7	1.1	4.9
SIEB	North Africa	1.1	1.0	0.8	1.3	...	4.3	0.4	1.0	1.3	3.0
BOYE	North Africa	0.8	2.3	1.2	...	...	...	...	...	...	...
BOUW	Eastern Europe	2.1	1.2	2.1	1.9	0.0	7.2	0.5	2.9	3.4	3.7
GREE	Eastern Europe	1.3	1.2	2.0	2.3	0.3	7.1	1.0	1.7	2.7	4.4
SIEB	Eastern Europe	1.8	2.1	1.8	2.5	...	8.2	0.8	4.8	5.7	2.6
BOYE	Eastern Europe	1.6	1.2	2.0	...	...	...	...	...	...	...
BOUW	Former USSR	10.1	10.5	2.6	6.4	0.0	29.6	1.6	7.7	9.3	20.3
GREE	Former USSR	5.6	9.0	2.6	8.5	0.6	26.3	3.2	6.2	9.4	16.9
SIEB	Former USSR	6.7	9.1	4.5	7.9	...	28.2	2.7	10.2	12.9	15.5
BOYE	Former USSR	6.7	16.4	2.6	...	...	...	...	...	...	...
BOUW	East Asia	12.3	8.5	24.4	10.9	0.0	56.1	7.8	23.9	31.6	24.5
GREE	East Asia	5.9	7.9	24.3	11.8	4.5	54.3	6.3	14.3	20.6	33.7
SIEB	East Asia	11.9	6.8	21.5	14.4	...	54.6	9.8	14.9	24.7	30.1
BOYE	East Asia	9.7	10.1	23.9	...	...	...	...	...	...	...
BOUW	Oceania	1.6	7.1	2.0	3.4	0.0	14.1	0.8	4.4	5.2	8.9
GREE	Oceania	0.7	7.9	0.8	3.1	0.0	12.6	1.1	2.1	3.2	9.4
SIEB	Oceania	1.2	10.4	0.6	4.0	...	16.3	1.1	2.1	3.2	13.1
BOYE	Oceania	0.9	8.8	0.8	...	...	...	...	...	...	...

[a]Units are N in Tg yr^{-1}.
[b]Numbers for *tm* may differ from those in Table 2 because here the original land mask of each data set was used here in combination with that of *IMAGE-team* [2001]. Data in Table 2 are based on the common land mask.

[33] Both models are lumped in space (0.5° by 0.5° grid cell) and time (annual time step) and produce estimates of total N (N_{tot}) in surface water from point sources and nonpoint sources. Total N includes nitrate, nitrite, ammonia, and dissolved and particulate organic N. We ignore the inputs from point sources to surface water. We use a meta version of the *Van Drecht et al.* [2003] model, representing the transport efficiency, which is the ratio of the N_{tot} export at the river mouth: *tm*. For the *Green et al.* [2004] model, we use reported transport efficiencies. The models are compared for continents and receiving oceans. The meta version of the *Van Drecht et al.* [2003] model is also used to compute river N export for the three data sets of N_{sur} (BOUW, GREE, and SIEB).

3. Results and Discussion

3.1. Global Scale

[34] The distribution of N_{sur} for the BOUW, GREE, and SIEB data sets suggests that the global and continental patterns differ (Figure 1), while the global calculated sums of N_{sur} (*tm*) for the three data sets are similar (Figure 2; Table 2). The matching mass *m* of N_{sur} is between 73 and 75%, indicating a fairly good agreement between the three data sets at the global scale (Table 2). This is confirmed by the *r* values for the three one-to-one comparisons with values of 0.73 to 0.75. However, there are major differences in the estimates of inputs and outputs.

[35] Perhaps the major disagreement is that both the sum of all N inputs and that of N outputs of the GREE data set are about 20% lower than the corresponding sums of the BOUW and SIEB data, and this reflects lower estimates for N_{dep} and N_{fix} for the inputs and N_{exp} and N_{vol} for the output terms (Table 3). There are also important differences in individual terms between BOUW and SIEB. N_{dep} of SIEB is 15% lower than BOUW, while N_{fix} of SIEB exceeds that of BOUW by 16% (Table 3).

[36] The BOUW data indicate higher N_{sur} values than the other two data sets in northern latitudes, which is caused by higher estimates for N deposition rates in these regions. The atmospheric N deposition rates used by BOUW are consistently higher than those in GREE (Figure 3). For most parts of the world the BOUW N_{dep} estimates also exceed those of BOYE and SIEB.

[37] A remarkable feature is that there is disagreement about atmospheric N deposition between BOYE and SIEB (Table 3, Figure 3), even though the same original data were used (Table 1). These differences are caused by smoothing

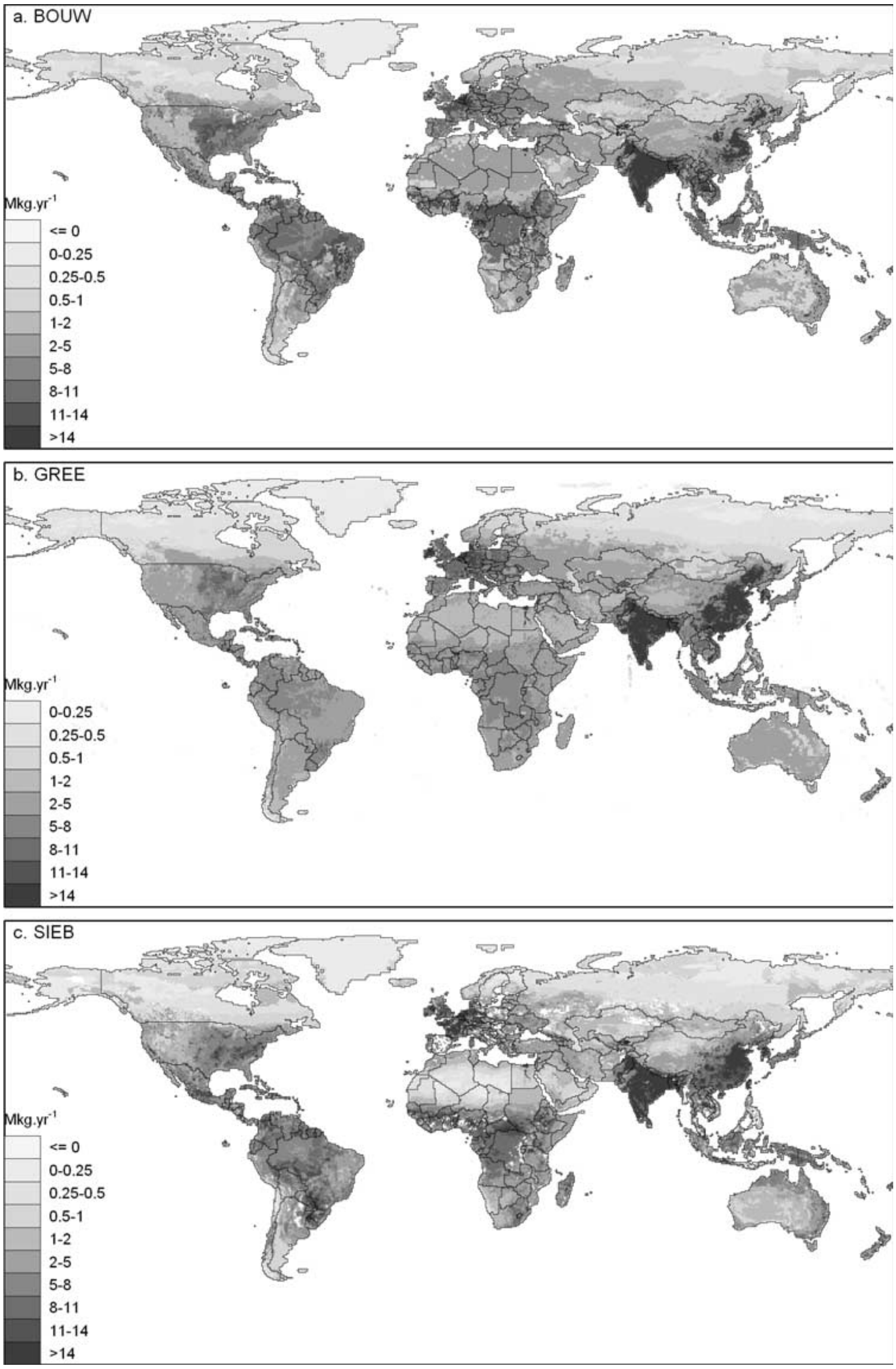

Figure 1. Global spatial distribution of N_{sur} for the (a) BOUW, (b) GREE, and (c) SIEB data sets.

of the spatial patterns in the maps (by SIEB and not in BOYE). Further differences arise from the representation of short-range transport (dry deposition) of NH_x in BOUW and SIEB, while this process is not included in BOYE and GREE (Table 1).

[38] The SIEB and GREE data indicate much lower N_{sur} values than BOUW in desert regions in Africa, Asia, and Australia (Figure 1). Low values of N_{sur} for deserts in SIEB and GREEN are caused mainly by the calculation routine of N_{fix} based on evapo-transpiration (SIEB)

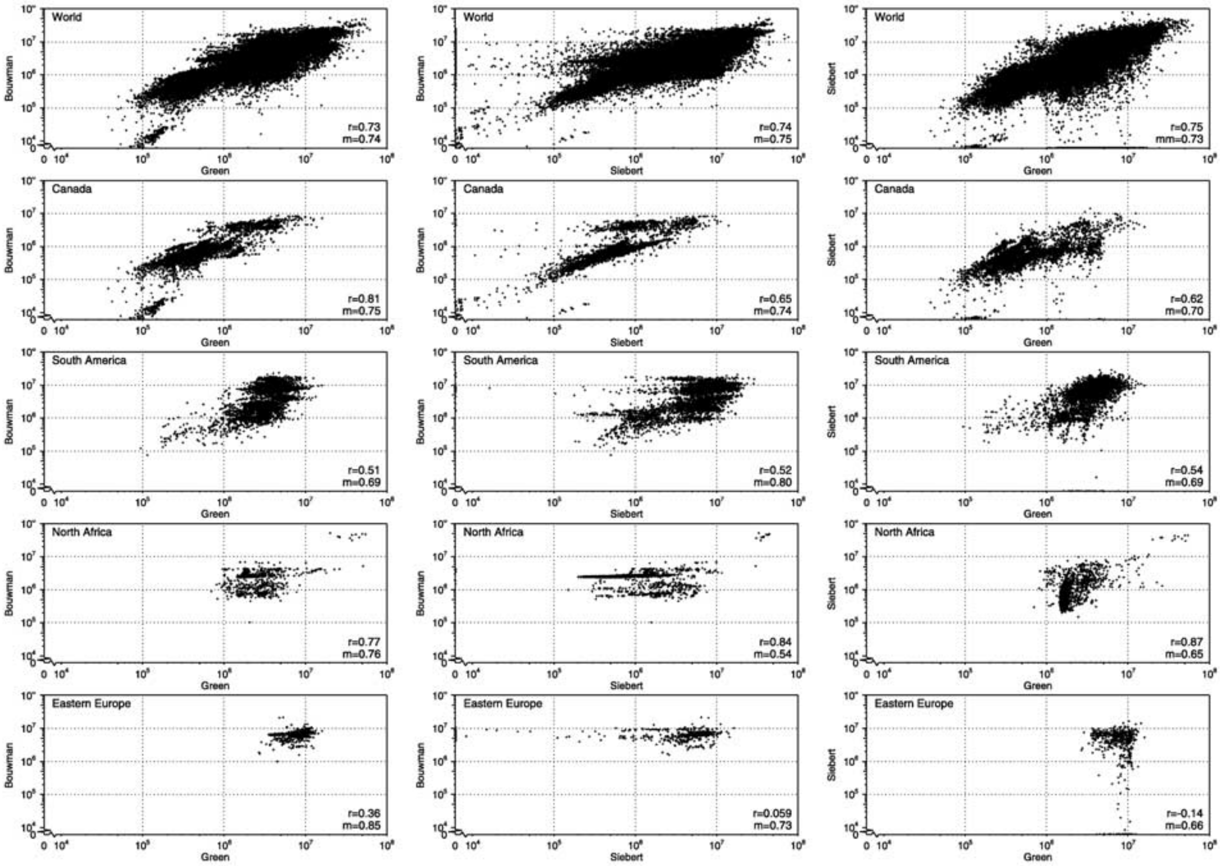

Figure 2. Comparison of grid values of N_{sur} for (left) GREEN (x) and BOUW (y), (middle) SIEB (x) and BOUW (y), and (right) GREEN (x) and SIEB (y) for the world and selected world regions. Both horizontal and vertical axes have a logarithmic scale.

and potential rates (GREE). BOUW is based on ecosystem-specific median values for different ecosystems and ignoring spatial differences in climate and net primary production in BOUW leads to discrepancies with the other two data sets (Figure 4).

[39] The differences in N fertilizer use between SIEB and BOUW on the one hand and BOYE and GREE on the other hand are related to differences in the statistical data used. The GREE and BOYE data sets use country data on total N fertilizer use (Table 1). In contrast, BOUW (country data)

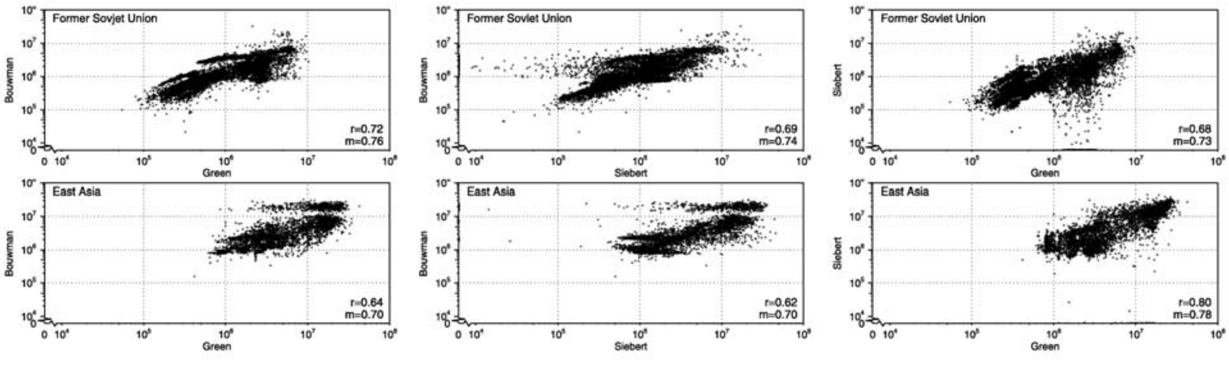

Figure 2. (continued)

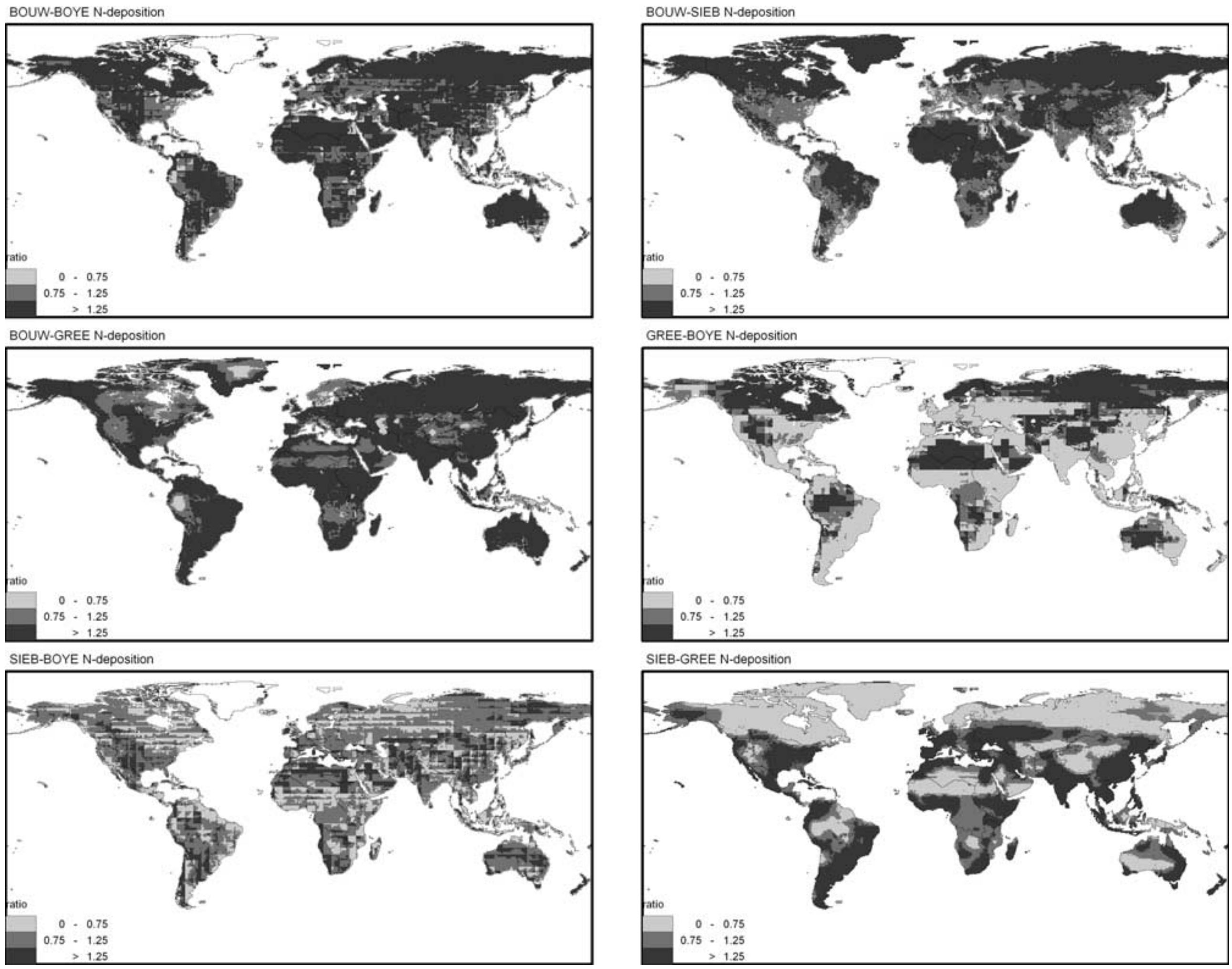

Figure 3. Ratio (top left) BOUW-BOYE, (top right) BOUW-SIEB, (middle left) BOUW-GREE, (middle right) GREE-BOYE, (bottom left) SIEB-BOYE, and (bottom right) SIEB-GREE for inputs from atmospheric N deposition.

and SIEB (aggregates for 17 world regions) use the expert estimates for fertilizer use by crop from *Food and Agriculture Organization of the United Nations* (*FAO*)/*International Fertilizer Industry Association* (*IFA*)/*International Fertilizer Development Center* (*IFDC*) [2003] for about 90 countries. BOUW complemented these data with the N fertilizer use by country of *FAO* [2001].

[40] The country totals for N fertilizer use calculated from *FAO/IFA/IFDC* [2003] may for some countries disagree with the country statistics of *FAO* [2003], and the combination of the estimated N application rates with crop acreages from a different source may create further discrepancies. For example, the N fertilizer use of SIEB for the former USSR exceeds that given by *FAO* [2003], and the global total N fertilizer use is underestimated by SIEB. For North Africa, both SIEB and BOUW underestimate N fertilizer use compared to FAO statistics [*FAO*, 2001, 2003]. It should be noted that there is also disagreement between BOYE and GREE (both based on data for countries) about N fertilizer use, for example for South America (Table 3).

[41] It is clear that another part of the differences in agricultural N inputs are caused by the land cover distributions used. The distribution of arable land used by BOUW to allocate N fertilizers and animal manure application is much more concentrated in smaller areas than in the other data sets (BOYE, GREE and SIEB) (Figure 5). The GREE and BOYE maps for fertilizer inputs are almost completely identical, except for a few areas such as in South America. Likewise, the way grassland is distributed also differs, and the animal manure distribution shows less spatial concentration in the SIEB and GREE than in the BOUW data sets (not shown).

[42] The estimated global sum of N_{exp} of SIEB and BOUW are similar, while GREE is much lower. The estimation of N export in harvested crop products is based on statistical information on crop production; hence differences are either caused by not taking into account the full range of agricultural crops, or by differences in the assumed N content of the harvested parts. The N uptake by animals during grazing in the GREE data is probably also lower than

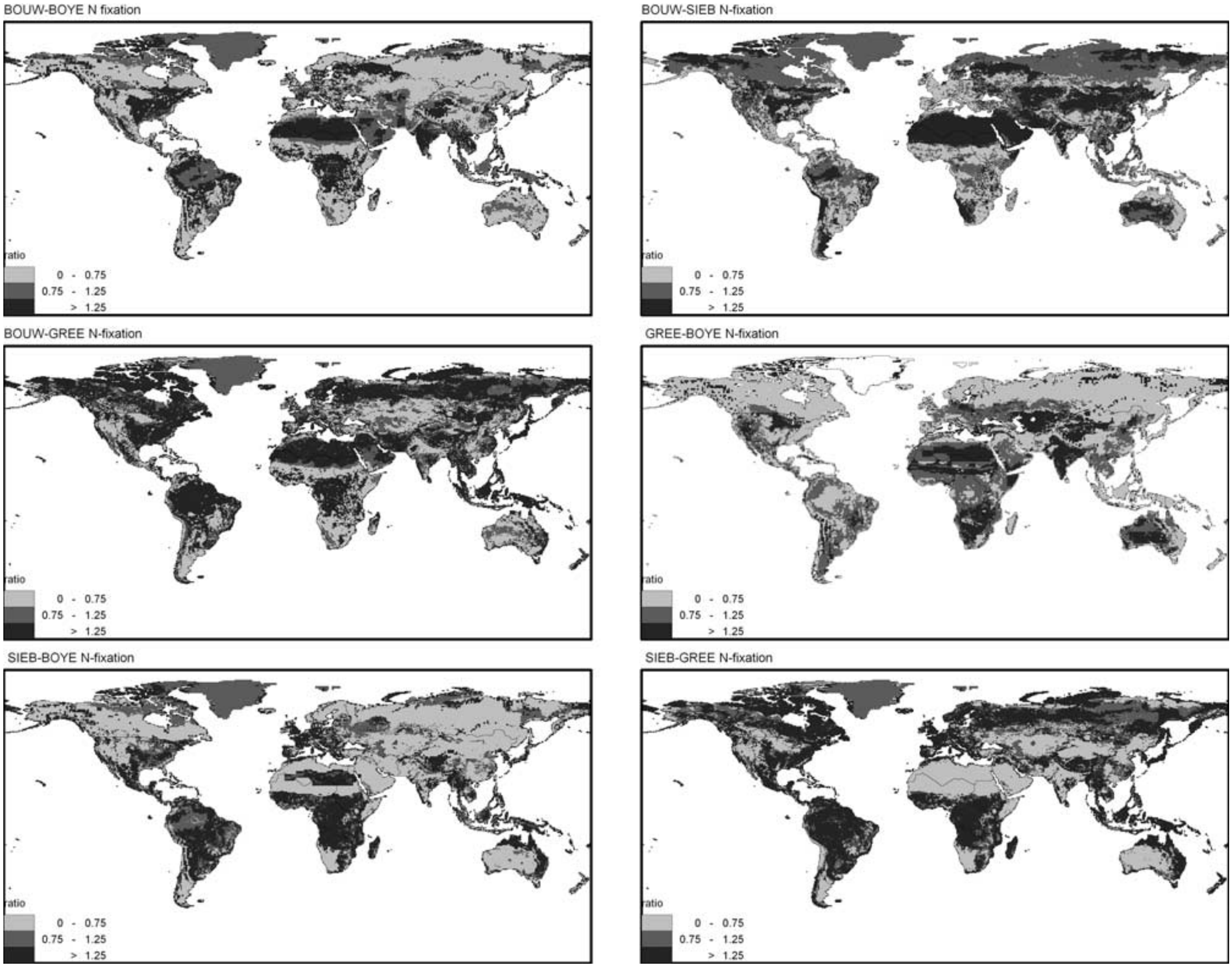

Figure 4. Ratio (top left) BOUW-BOYE, (top right) BOUW-SIEB, (middle left) BOUW-GREE, (middle right) GREE-BOYE, (bottom left) SIEB-BOYE, and (bottom right) SIEB-GREE for N inputs from biological N fixation.

for SIEB and BOUW, since the N excretion rates of GREE are also considerably lower.

[43] With lower values for N excretion rates by the various animal categories, it is not surprising that the global N_{vol} estimate of GREE is also lower than BOUW and SIEB. The global estimate of SIEB for N_{vol} exceeds that of BOUW by 30%. This is because BOUW did not include emissions from stored manure in their surface N balance approach, while SIEB presents the NH_3 volatilization for the livestock production as a whole.

3.2. Continents and World Regions

[44] There is much stronger disagreement between the data sets at the scale of world regions than at the global scale (Figure 2; Table 2). The causes of the differences in N_{dep}, N_{fix}, N_{fert}, N_{exp}, and N_{vol} between the data sets at the global scale were discussed in section 3.1. Here we will discuss other remarkable differences which become apparent when comparing world regions, taking Canada, South America, Eastern Europe, and the former USSR, East Asia, and North Africa as examples (Table 3).

[45] Starting with Canada, we see that *tm* according to BOUW is 50% higher than that of GREE and SIEB (Table 3). This is primarily due to the estimates for biological N fixation and atmospheric N deposition in BOUW, which exceed those of SIEB and GREE. Since the output terms of BOUW and SIEB are in reasonable agreement, this causes a higher N_{sur} in BOUW. The differences between GREE and the other data sets for Canada reflect the differences we already noted for the global-scale comparison. The BOYE estimate for biological N fixation for Canada exceeds those of the other three data sets by a factor of 1.4 to 2.9.

[46] For South America, *tm* according to GREE is much lower than for the other two data sets (Table 3). While there is a good general agreement at the level of total N inputs and total outputs between BOUW and SIEB (Table 3), all input terms in the GREE data are much lower than in the other data sets. All four data sets agree that the N_{fix} term is

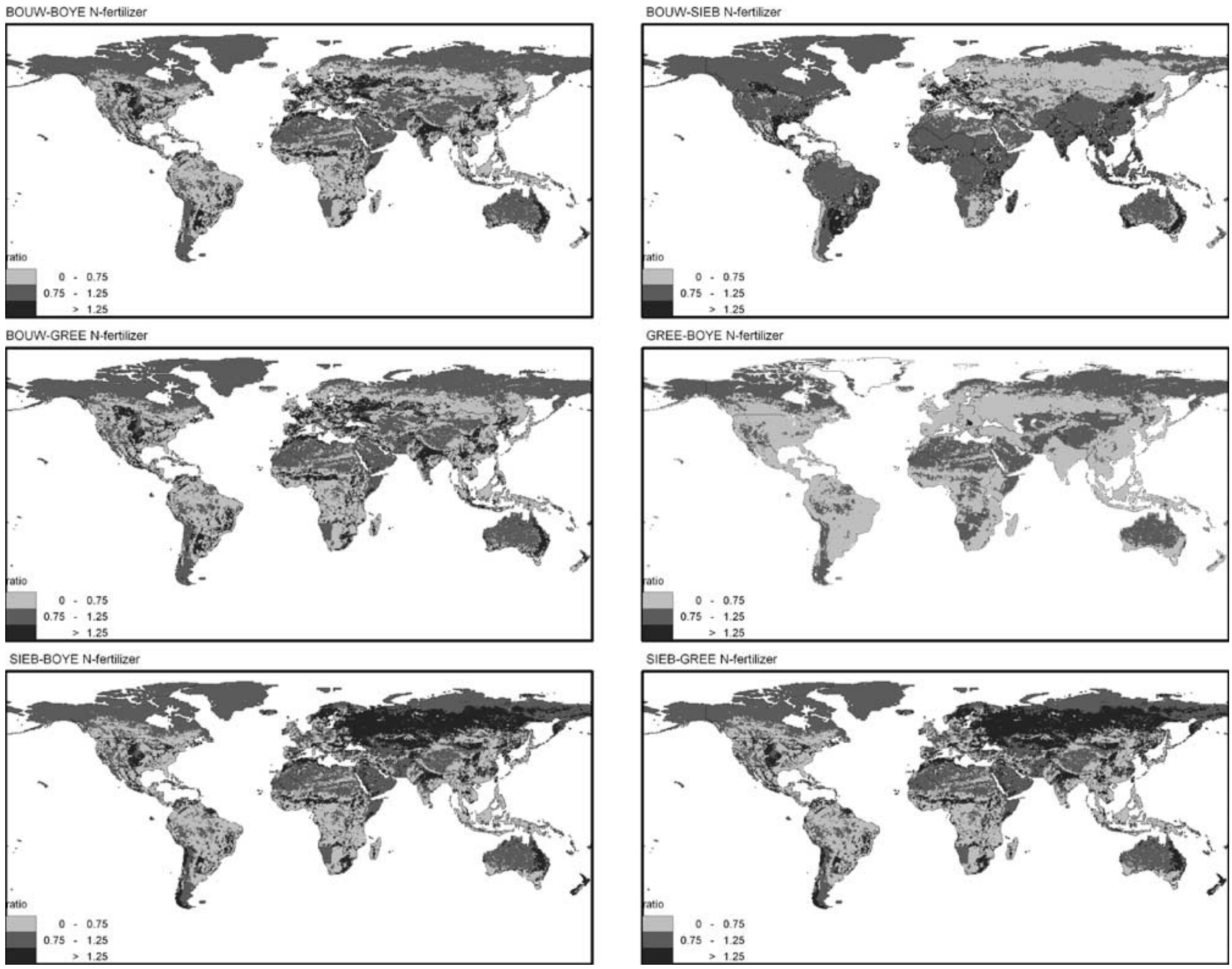

Figure 5. Ratio (top left) BOUW-BOYE, (top right) BOUW-SIEB, (middle left) BOUW-GREE, (middle right) GREE-BOYE, (bottom left) SIEB-BOYE, and (bottom right) SIEB-GREE for fertilizer N inputs.

the most important contributor to N inputs in South America, but the estimates of BOYE and especially GREE are far lower than the other data sets.

[47] The sum of N_{sur} (*tm*) provided by SIEB for North Africa is much lower than GREE (60%) and BOUW (48%) (Table 3), and this is mainly caused by N_{fix}, which is 50% of the BOYE estimate, 35% of the GREE estimate, and 25% of BOUW. The BOUW estimate for N_{exp} exceeds those of the GREE and SIEB data.

[48] Turning to Eastern Europe, we note that *tm* according to SIEB is much lower than for the other two data sets (Table 3). While the SIEB total N inputs for Eastern Europe exceed those of BOUW (+14%) and GREE (+15%), the SIEB total outputs exceed those of BOUW by 65% and those of GREE by 114% leading to a 29% lower N_{sur} than in BOUW and 41% lower than in GREE. The SIEB estimate for biological N fixation is much higher than the other three data sets.

[49] There is major disagreement about *tm* for the former USSR between SIEB and GREE on the one hand and BOUW on the other hand (Table 3). While the sum of N inputs is similar in all three data sets, the outputs are different. The major difference is in the N_{exp}, which is much higher in the SIEB data than in BOUW and GREE. Further striking differences are in the estimates for N_{dep} (highest in BOUW, as discussed in section 3.1), N_{fert} (highest in SIEB), and N_{fix} (BOYE exceeds the other data sets by a factor of 1.6–1.8).

[50] While the input terms for East Asia show a good overall agreement between the data sets, *tm* according to GREE for East Asia exceeds the estimates of BOUW (38%) and SIEB (12%) (Table 3). The main difference is in the N export by crops, grass, and fodder, where BOUW exceeds SIEB by 60% and GREE by 67%. It is difficult to determine what causes these differences. However, an independent estimate for N export in crops (excluding N grazing, fruit trees, mulberry, and tea) of 15.3 Tg yr^{-1} for 1998 for China [*Zhu and Chen*, 2002], the most important country in the East Asia region, suggests that GREE and SIEB N_{exp} for East Asia are probably underestimates. The SIEB estimate

for N_{fix} is lowest and that of BOYE highest (exceeding SIEB by almost 50%).

[51] One major output term of the surface balance is the crop export N_{exp}. As discussed for the global scale, the N_{exp} estimates of GREE are low for most world regions compared to BOUW and SIEB. The SIEB data set is based on simulated yields for each grid cell (Table 1), which are scaled using a regional calibration factor to obtain the regional production given by *FAO* [2003]. Crop yields tend to be overestimated in zones of low productivity and underestimated in zones of high productivity within the same world region. Since SIEB considers country data for fertilizer N inputs and country or even subnational data for animal manure inputs, N_{sur} may be overestimated in countries with high productivity relative to the regional mean and underestimated in countries with low relative productivity. The same problem occurs in the SIEB data as a result of using regional average cropping intensities. This may explain the differences between SIEB and BOUW for East Asia, Eastern Europe and the former USSR. It should be noted that similar scale problems occur within countries in the BOUW and GREE data by using country data for inputs and crop yields, which do not reflect heterogeneity in agricultural systems.

[52] In the SIEB data, about 1% of the grid cells have zero N_{sur} values (originally negative, i.e., soil N depletion, see Figure 1) in many parts of the world, for example, Canada, Argentina, Spain, Eastern European countries, Russia, and Khazakstan (Figure 1 and 2). In addition, N_{sur} values in several world regions are much more variable according to SIEB than for the other two data sets. This is most clearly reflected by clusters with similar values for BOUW or GREE while SIEB gives a wide range, particularly in North Africa and Eastern Europe (Figure 2) and United States, Western Africa, Western Europe, and Southeast Asia (not shown). The higher variability is caused by higher variability of some N inputs (manure, biological N-fixation) and probably by the calculated changes in soil N pools in the SIEB data. This aspect was not addressed in the compilation of the GREE data. In the BOUW data set the different crops within each grid cell were considered to represent a rotating system, where the surplus of one crop may serve to compensate for the deficit of the next crop thus portraying the long-term balance for larger areas. Both the assumed zero N_{sur} values and the dynamic calculation of the soil N pools probably cause low *r* values in many world regions for the BOUW-SIEB and GREE-SIEB comparisons.

[53] The above examples indicate that there may be apparent agreement in the regional sum of N_{sur} (*tm*), while the underlying data may strongly disagree. This is also illustrated by the matching mass of N_{sur}, *m*, which varies between 64% (Oceania) and 88% (South Asia) for the BOUW/GREE comparison, 60% (North Africa) and 84% (South Asia) for GREE/SIEB, and 49% (North Africa) to 86% (South Asia) for BOUW/SIEB (Table 2; Figure 2).

[54] The Pearson correlation coefficient (*r*) is much more variable than *m*. One of the causes is the distribution of the estimates. For a distribution with one cluster of estimates, the *r* value is inherently low, while the *m* may be much higher, indicating a good agreement. For a one-to-one comparison where values in both *x* and *y* direction cover a wider range, the *r* value will be higher than for small clusters.

[55] The BOUW/GREE comparison yielded low *r* values of 0.1–0.4 for Central America, Eastern Europe, Eastern and Southern Africa, Southeast Asia, and Oceania. The GREE/SIEB comparison gave low *r* values for the same regions, and even a negative value for Eastern Europe. The BOUW/SIEB comparison resulted in higher values for these regions (0.3–0.4), except for Eastern Europe with an *r* of <0.1.

3.3. Large Countries

[56] The results for large countries provide more detailed insight in the differences between the data sets. Canada and the United States (Table 2) have *m* of 0.7–0.8 for the BOUW/GREE and BOUW/SIEB, and somewhat lower for the GREE/SIEB comparison. For China the *m* value exceeds 0.7 for the three one-to-one comparisons, while results are more variable for Brazil (0.68–0.79) than for India (0.86–0.90).

[57] The values for *r* for Canada, United States, and China range from 0.6 to 0.8. The *r* values exceed 0.6 for India and China, while r for Brazil is <0.1 for the BOUW/GREE and BOUW/SIEB comparison. The results for Brazil (8.4 Mkm^2) as a whole are surprising, since the *r* values for the Amazon river basin (5.7 Mkm^2) are much higher (see below). Apparently, the disagreement between BOUW on the one hand and GREE and SIEB on the other hand are caused by differences in Brazil outside the Amazon River basin. This is a consequence of large differences in the allocation of N fertilizer (Figure 5) and animal manure N. For N inputs from animal manure the differences are clearly related to the difference of the basic data used, which is country scale for BOUW and subnational scale for SIEB. For fertilizer N inputs it is not clear what causes the differences, since both SIEB and BOUW rely on the same basic land-cover data (Table 1).

3.4. River Basins

[58] Since the purpose of this comparison is to estimate the riverine N export to coastal marine systems, we also compared the estimates of the different data sets for river basins (Table 4). When looking at the data for the 25 largest river basins (varying in size from 5.78 for the Amazon to 0.84 million km^2 for the Senegal) we see that there is disagreement about the *tm* between GREE on the one hand and BOUW and SIEB for the Amazon and Parana. For these river basins, GREE has lower *tm* values similar to Brazil as a whole. Furthermore, the agreement between the three data sets is fairly constant, *m* being of the order of 70% with a few exceptions.

[59] The comparison indicates that the *r* values for the Amazon are higher than for the Parana and Orinoco, which is caused by a stronger degree of clustering in the latter two rivers and less so by lack of agreement. The comparison also yields low *r* values (<0.3) for the Zaire and Zambezi (BOUW/GREE and GREE/SIEB), Senegal (BOUW/GREE), Volga, Murray, and Orange (for all one-to-one comparisons).

Table 4. River Basin Area, Sum of N_{sur} (*tm*) for BOUW, GREE, and SIEB, and the Fraction of Total Mass of N With Match (*m*), and Pearson Correlation Coefficient (*r*) for the Three One-by-One Comparison of the Data Sets for the 25 Largest River Basins of the World Using Data From *Fekete et al.* [2002]

River Basin	Area, 10^6 Km^2	BOUW *tm*, Tg yr^{-1}	GREE *tm*, Tg yr^{-1}	SIEB *tm*, Tg yr^{-1}	BOUW/GREE		BOUW/SIEB		GREE/SIEB	
					m	*r*	*m*	*r*	*m*	*r*
Amazon	5.8	15.0	9.1	15.8	0.73	0.45	0.86	0.41	0.72	0.22
Nile	3.7	6.7	5.0	8.2	0.68	0.47	0.72	0.70	0.61	0.47
Zaire	3.6	10.7	6.8	12.3	0.70	0.04	0.88	0.59	0.68	0.06
Mississippi	3.2	10.7	8.2	9.2	0.81	0.59	0.80	0.50	0.81	0.47
Amur	2.9	4.4	4.3	2.8	0.72	0.73	0.61	0.33	0.67	0.47
Parana	2.6	5.9	3.7	7.5	0.68	0.29	0.72	0.25	0.62	0.35
Yenisei	2.6	2.2	2.0	1.2	0.79	0.42	0.61	0.23	0.64	0.28
Ob	2.6	1.3	1.0	1.1	0.79	0.59	0.74	0.38	0.78	0.38
Lena	2.4	1.0	0.6	0.8	0.77	0.52	0.82	0.90	0.85	0.32
Niger	2.2	3.5	3.4	4.3	0.70	0.24	0.68	0.74	0.59	0.36
Zambezi	1.9	3.0	3.3	4.4	0.63	0.27	0.69	0.48	0.78	0.18
Chang Jiang	1.8	6.1	8.4	8.4	0.71	0.46	0.75	0.61	0.85	0.70
Mackenzie	1.6	7.9	8.7	9.8	0.89	0.90	0.85	0.79	0.83	0.75
Ganges	1.6	0.7	0.5	0.5	0.75	0.59	0.77	0.51	0.73	0.36
Chari	1.6	2.0	2.0	2.3	0.68	0.35	0.63	0.60	0.67	0.74
Volga	1.4	2.4	1.8	1.7	0.75	−0.29	0.71	0.06	0.72	−0.04
St. Lawrence	1.1	3.8	4.1	5.0	0.84	0.81	0.82	0.83	0.82	0.79
Indus	1.0	2.8	1.8	3.2	0.70	−0.14	0.87	0.25	0.71	0.01
Nelson	1.0	1.5	1.0	1.7	0.59	0.08	0.66	0.22	0.62	−0.22
Orinoco	1.0	2.5	2.0	1.8	0.76	0.62	0.76	0.71	0.81	0.71
Murray	1.0	1.7	1.6	0.7	0.86	0.80	0.54	0.52	0.56	0.54
Orange	0.9	0.8	1.2	1.4	0.71	−0.24	0.55	0.13	0.61	−0.24
Huang He	0.9	2.7	3.1	3.0	0.69	0.52	0.71	0.59	0.80	0.76
Yukon	0.8	0.4	0.2	0.3	0.68	0.31	0.83	0.83	0.75	0.29
Senegal	0.8	0.8	1.1	1.0	0.59	0.09	0.58	0.49	0.65	0.56

[60] It is clear that the agreement is much better for large than for small river basins (Figure 6). This is caused by the increasing uncertainty of spatial distributions of land use and management along with decreasing area.

3.5. Comparison of River N Export Models

[61] The transport coefficients of *Green et al.* [2004] are the result of calibration against data for about 50 rivers. The *Van Drecht et al.* [2003] model is not calibrated. The two river N export models gave comparable results when all data are aggregated to the global scale, with a transport efficiency of 18% (Table 5). However, at the scale of continents we see that there is good agreement for industrialized regions (North America, Europe, Australia). For the developing countries, there are more pronounced differences, whereby in some parts of the world the *Green et al.* [2004] model predicts a somewhat higher transport efficiency (Asia, South America), and for other parts it yields a lower efficiency (Africa). The transport efficiencies are in good agreement at the scale of receiving oceans (Table 5). For countries and individual river basins, there will be more disagreement, since the average river basin temperature is the only variable in the *Green et al.* [2004] model, while that of *Van Drecht et al.* [2003] uses specific information for individual grid cells.

[62] The three data sets for N_{sur} were used with the transport efficiency for nonpoint sources of the meta model derived from the *Van Drecht et al.* [2003] model. Using this

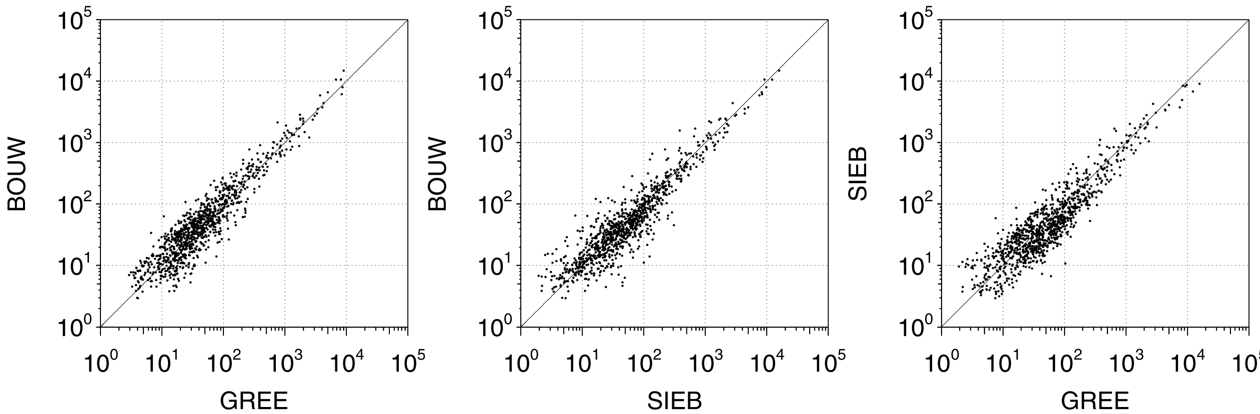

Figure 6. Comparison of *tm* for the 1000 largest river basins for (left) BOUW-GREE, (right) BOUW-SIEB, and (right) SIEB-GREE.

Table 5. Transport Efficiency for Continents and Receiving Oceans Derived From Two Models for Calculating River N Export[a]

Region/Ocean	*Van Drecht et al.* [2003][b] Nonpoint Sources	*Van Drecht et al.* [2003][b] Overall	*Green et al.* [2004] Overall
Africa	16	16	6
Asia	16	17	21
Australia	4	4	4
Europe	22	25	24
North America	17	19	16
South America	23	24	26
World	18	19	18
Arctic Ocean	22	22	22
Atlantic Ocean	21	22	21
Indian Ocean	13	14	13
Mediterranean+Black Sea	12	14	12
Pacific Ocean	21	22	23

[a]Transport efficiency is the ratio of modeled river N export: *tm* (including point sources).
[b]Transport efficiency for nonpoint sources is that for *tm* presented in Table 2, and is used to compute the river export of N from nonpoint sources (see Table 6). The overall transport efficiency is calculated for the nonpoint and point sources as presented by *Bouwman et al.* [2005a] and can be compared with the overall transport efficiency of *Green et al.* [2004].

model, there is a clear difference between the three data sets on the global scale (Table 6), the estimate based on GREE being 16% lower than on the basis of BOUW, while river export based on SIEB exceeds BOUW by 10%.

[63] SIEB results in higher values than the BOUW data set in all world regions except for Europe and North America. The largest difference is seen in Australia where the river export for the SIEB data exceeds that on the basis of BOUW by 76%. Since *tm* of the SIEB data for Australia exceeds that of BOUW by 46%, there is a clear effect of differences in the spatial allocation of the nonpoint sources. While the estimates for N deposition in the BOUW data set exceed those in GREE and SIEB (Figure 3), most of it comes down in dry regions where there is no or negligible downward transport of water and nitrate. Conversely, the N surplus in the north and east of Australia (Figure 1) (with the SIEB N fixation exceeds that of BOUW and SIEB, Figure 4) gives rise to more river N export according to SIEB than for BOUW and GREE. Similar effects of differences in the spatial distribution of N_{sur} occur in Africa, where differences between SIEB and BOUW and GREE and BOUW in the calculated river N export are much larger than those in the total N_{sur}. The BOUW data for *tm* for the Sahara exceed those of GREE and SIEB due to higher N deposition and biological N fixation rates. However, in many other more humid parts of Africa particularly the SIEB data set indicates higher N deposition rates, N fixation, and N fertilizer use than BOUW.

[64] River N export stemming from nonpoint sources calculated for the GREE data are lower than those calculated with BOUW for all world regions, except for Asia. The largest difference is seen in South America, where the calculated river export with the GREE data is only 63% of that calculated with BOUW. This reflects the difference in the *tm*, and suggests that in areas with humid climates differences in the spatial allocation of N_{sur} are less important than in dry climates.

[65] The comparison of the river N export from N stemming from nonpoint sources clearly shows where differences in *tm* and its spatial distribution occur; that is, the main disagreement is in Africa, Australia, South America, and North America. The SIEB estimates for *tm* tend to be highest in all continents, while the BOUW data tend to be highest in regions discharging to the Arctic Ocean, mainly the result of higher estimates of atmospheric N deposition rates as discussed in section 3.1.

4. Conclusions and Recommendations

[66] We conclude that there are important differences between the data sets which are compared in this study at the global, regional, country, and river-basin scales. There are differences in the surface balance for N, individual N input and output terms, and the spatial distributions. The most uncertain aspects of the surface balance are the input

Table 6. Area and *tm* for BOUW, GREE, and SIEB, and River N Export Calculated With the Transport Coefficient for Nonpoint Sources From Table 5 From the Model of *Van Drecht et al.* [2003] for the BOUW Data Set and River N Export for GREE and SIEB Relative to BOUW for Continents and Receiving Oceans

Region/Ocean	Area, Mha	BOUW *tm*, N in Tg yr^{-1}	GREE *tm*[a]	SIEB *tm*[a]	River N export: BOUW, N in Tg yr^{-1}	River N export: GREEN[a]	River N export: SIEB[a]
Africa	29	48	84	118	8	76	124
Asia	41	71	112	110	12	111	118
Australia	7	7	114	146	0	86	176
Europe	10	25	96	89	6	91	86
North America	21	34	81	97	6	76	93
South America	17	40	63	112	9	63	112
World	128	232	91	109	42	84	110
Arctic Ocean	16	10	67	68	2	64	67
Atlantic Ocean	44	100	76	106	21	72	105
Indian Ocean	20	43	106	129	6	97	130
Mediterranean+Black Sea	11	20	91	96	2	96	107
Pacific Ocean	20	43	108	117	9	107	120

[a]Relative to BOUW (BOUW = 100).

terms biological N fixation and atmospheric N deposition, and the output terms (export of N from fields in harvested products and grazing, and NH_3 volatilization).

[67] The differences between the data sets in the estimates for N inputs from natural biological N fixation rates are much larger than those for the managed agricultural inputs (fertilizer, animal manure). Our understanding of spatial patterns and rates of biological N fixation is better in agricultural systems than in natural ecosystems [*Galloway et al.*, 2004]. In agricultural lands, we have relatively good records of the distribution of cultivated crop lands and a long history of measurements of rates of symbiotic N fixation in cultivars, along with statistical information on agricultural management practices [*Smil*, 2001].

[68] However, in natural lands, it remains a challenge even to map the spatial distribution of natural vegetation species hosting N fixing bacteria, such as alder shrub species that cover only small fractions of forested wetland environments [*Boyer et al.*, 2002]. There is a broad spectrum of N fixing organisms in the natural environment, which have complex distributions across the landscape. Furthermore, even in a single plant community, there is a large variability in the distribution of observed fixation rates, and there is a large degree of spatial and temporal heterogeneity in factors controlling fixation rates [*Smil*, 2001].

[69] All four data sets described herein for natural fixation make use a recent compilation of biological N fixation based on published literature values in natural ecosystems [*Cleveland et al.*, 1999]. Hence any differences in the estimates of natural biological N fixation illustrated here stem largely from the way these data were interpreted and scaled by coupling to other ecosystem process models (see Table 1). The review by *Cleveland et al.* [1999] provides a good starting point for considering the magnitude of biological N fixation at regional scales, though the estimates are still poorly constrained owing to a relatively sparse number of measurements and a large degree of variability in observed rates. There may also be biases in literature values given the fact that rate measurements are typically made in individual plots where there are large assemblages of N fixing species, which in turn might not be representative of an entire ecoregion (C. Cleveland and G. Asner, personal communication, 2002). Thus there remains huge uncertainty in understanding the magnitude of natural biological N fixation at regional scales, highlighting the need for considerably more research in this area.

[70] The major cause of difference between the estimates for atmospheric N deposition is the chemistry-transport model (CTM) and the N emission data used in the CTM run. First, the grid size of the CTMs used is comparable to that of a small to medium-sized river basin, and heterogeneity within the basin is only poorly described. Second, there may have been inconsistencies between the emission input data used to calculate N deposition, and the data set for which the calculations were done.

[71] Although total global, regional, and national N inputs from N fertilizer and animal manure show less differences between the data sets than biological N fixation and atmospheric N deposition, the differences in the spatial allocation of N fertilizer and animal manure inputs between the data sets indicate that this aspect is also poorly known. In addition, N surpluses in agricultural systems are responsible for the major part of the global total N surplus, and agricultural systems are concentrated in a relatively minor part of the terrestrial land area (about 15% for intensively managed agricultural systems, and 30–40% for total agriculture including extensively used and marginal lands).

[72] The NH_3 volatilization from N fertilizer and animal manure are all calculated with comparable methods, and main differences are in the spatial allocation of fertilizers and animal manure. The N removal from agricultural fields by harvesting and grazing is estimated by assigning fixed N contents to crop production or yield data. The major differences are caused by not taking into account the full range of agricultural crops, or by differences in the assumed N contents.

[73] A number of recommendations for future improvements can be given on the basis of our comparison. Regarding biological N fixation in natural ecosystems the BOYE approach using an ecosystem model seems to be the most promising, and future research directions in this respect are given by *Vitousek et al.* [2002].

[74] Modeling rates and spatial distribution of atmospheric N deposition remains a major challenge, reflecting the complexities of a multitude of different sources, transport pathways, chemical transformations, and parameterizations of deposition within the model structures [*Dentener and Crutzen*, 1994; *Prospero et al.*, 1996; *Holland et al.*, 1999]. Moreover, there is incomplete monitoring of N species with which to calibrate the models, with relatively sparse data over space and time for wet deposition species, and with only a paucity of data on dry and organic components of deposition [*Meyers et al.*, 2001]. Regional-scale estimations of atmospheric N deposition may be improved by taking advantage of recent refinements in resolution and accuracy of CTMs, such as the newly developed TM5 model [*Krol et al.*, 2004]. The spatial resolution of TM5 (1°) is finer than that of the predecessor TM3 versions used in SIEB, BOYE, and GREE. Consistent model simulations of N deposition based on three-dimensional transport from atmospheric NH_3 and NO emissions sources are currently being evaluated, potentially enabling a better description of N deposition at regional and global scales.

[75] Better spatial allocation and description of land-use systems is important, since intensive crop and livestock production systems are generally concentrated in areas with good soils such as river floodplains. The climatic and hydrological characteristics of such regions may differ from those in less intensive areas.

[76] With respect to inputs and outputs in crop production systems, a major improvement can be achieved by using a crop growth model approach that better reflects the agricultural management on the country or subnational scale. By doing so the simulation of crop yields in response to the inputs of N and other nutrients and the crop N-uptake could be improved, as well as the biological N fixation by leguminous crops. A further improvement is the use of information on fertilizer use in irrigated and rainfed systems, in combination with available estimates of areas

where no fertilizer is applied at all, fallow, crop rotations, and regions with multiple cropping.

[77] Regarding animal manure, the SIEB approach provides detailed spatial information on animal densities. This approach can be improved by adding information on livestock production systems, including characteristics such as type and quality of animal feedstuffs including grazing, feed availability during the year, and excretion rates based on these production characteristics, animal manure management, and associated NH_3 volatilization.

[78] At the river-basin scale the different data sets agree fairly well for large river basins, although there is considerable disagreement about the grid-by-grid allocation within the river basins. Comparison of large-scale transport efficiencies of the two models compared [*Van Drecht et al.*, 2003; *Green et al.*, 2004] showed a good agreement for the global scale, most continents, and receiving oceans. The models are different with much lower transport efficiency for Africa for the *Green et al.* [2004] model than the *Van Drecht et al.* [2003]. Hence a conceptual model which is not calibrated [*Van Drecht et al.*, 2003] can yield results which are close to those obtained with a calibrated model [*Green et al.*, 2004].

[79] The river N export based on BOUW, GREE, and SIEB calculated with the meta version of *Van Drecht et al.* [2003] reflect differences in *tm*, except for regions or continents with vast areas with dry climates. In dry climates the downward flow of water is limited and differences in *tm* are less important than in more humid climates. However, at the scale of individual river basins the differences between the data sets are much larger than for continents or world regions.

[80] There is a tendency toward increasing the spatial and temporal detail in studies on global-scale changes in N cycling. However, data on other aspects such as agricultural management and N cycling in natural ecosystems are available at the scale of states or provinces at best. Hence smaller scales such as river basins or sub-basins may be more appropriate to test models. The most important challenge is to develop approaches to compare results at the model scale (1–50 km) with field measurements (plot scale).

[81] **Acknowledgments.** We gratefully acknowledge the support of the UNESCO Intergovernmental Oceanographic Committee for various workshops that were held which formed the basis for the contribution to the Global NEWS project described in this paper. The work of G. V. D. and A. F. B. was part of the project Integrated Terrestrial Modeling (S/550005/01/DD) of the Netherlands Environmental Assessment Agency, National Institute for Public Health and the Environment. E. W. B. and P. A. G. thank Greg Asner, Cory Cleveland, Frank Dentener, and Charlie Vörösmarty for their data contributions to our individual modeling efforts as detailed in prior publications cited herein.

References

Alexander, R. B., P. J. Johnes, E. W. Boyer, and R. A. Smith (2002), A comparison of models for estimating the riverine export of nitrogen from large watersheds, *Biogeochemistry*, *57/58*, 295–339.
Asner, G. P., A. R. Townsend, W. J. Riley, P. A. Matson, J. C. Neff, and C. C. Cleveland (2001), Physical and biogeochemical controls over terrestrial ecosystem responses to nitrogen deposition, *Biogeochemistry*, *54*, 1–39.
Bouwman, A. F., and H. Booij (1998), Global use and trade of feedstuffs and consequences for the nitrogen cycle, *Nutr. Cycl. Agroecosyst.*, *52*, 261–267.
Bouwman, A. F., D. S. Lee, W. A. H. Asman, F. J. Dentener, K. W. Van der Hoek, and J. G. J. Olivier (1997), A global high-resolution emission inventory for ammonia, *Global Biogeochem. Cycles*, *11*, 561–587.
Bouwman, A. F., R. G. Derwent, and F. J. Dentener (1999), Towards reliable global bottom-up estimates of temporal and spatial patterns of emissions of trace gases and aerosols from land-use related and natural sources, in *Scaling of Trace Gas Fluxes in Ecosystems*, edited by A. F. Bouwman, pp. 1–26, Elsevier, New York.
Bouwman, A. F., L. J. M. Boumans, and N. H. Batjes (2002a), Estimation of global NH_3 volatilization loss from synthetic fertilizers and animal manure applied to arable lands and grasslands, *Global Biogeochem. Cycles*, *16*(2), 1024, doi:10.1029/2000GB001389.
Bouwman, A. F., L. J. M. Boumans, and N. H. Batjes (2002b), Modeling global annual N_2O and NO emissions from fertilized fields, *Global Biogeochem. Cycles*, *16*(4), 1080, doi:10.1029/2001GB001812.
Bouwman, A. F., D. P. Van Vuuren, R. G. Derwent, and M. Posch (2002c), A global analysis of acidification and eutrophication of terrestrial ecosystems, *Water Air Soil Pollut.*, *141*, 349–382.
Bouwman, A. F., G. Van Drecht, J. M. Knoop, A. H. W. Beusen, and C. R. Meinardi (2005a), Exploring changes in river nitrogen export the world's oceans, *Global Biogeochem. Cycles*, *19*, GB1002, doi:10.1029/2004GB002314.
Bouwman, A. F., K. W. Van der Hoek, B. Eickhout, and I. Soenario (2005b), Exploring changes in world ruminant production systems, *Agric. Syst.*, *84*(2), doi:10.1016 j.agsy 2004.05.006.
Bouwman, A. F., G. Van Drecht, and K. W. Van der Hoek (2005c), Nitrogen surface balances in intensive agricultural production systems in different world regions for the period 1970–2030, *Pedosphere*, *15*(2), 137–155.
Boyer, E. W., C. L. Goodale, N. A. Jaworski, and R. W. Howarth (2002), Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern U.S.A., *Biogeochemistry*, *57/58*, 137–169.
Boyer, E. W., R. W. Howarth, J. N. Galloway, F. J. Dentener, C. Cleveland, G. P. Asner, P. Green, and C. Vörösmarty (2004), Current nitrogen inputs to world regions, in *Agriculture and the Nitrogen Cycle: Assessing the Impacts of Fertilizer Use on Food Production and the Environment*, edited by A. R. Mosier, J. K. Syers, and J. R. Freney, pp. 221–230, Island, Washington, D. C.
Brandjes, P. J., J. D. Wit, H. G. Van Der Meer, and H. Van Keulen (1996), Livestock and the environment—Finding a balance: Environmental impact of animal manure management, report, Livestock, Environ. and Dev. Initiative (LEAD) of FAO, Int. Agric. Cent. (IAC), Wageningen, Netherlands. (Available at http://www.fao.org/DOCREP/004/X6113E/x6113e00.htm)
Bruinsma, J. E. (2003), *World Agriculture: Towards 2015/2030—An FAO Perspective*, 432 pp., Earthscan, London.
Cleveland, C. C., et al. (1999), Global patterns of terrestrial biological nitrogen (N_2) fixation in natural ecosystems, *Global Biogeochem. Cycles*, *13*, 623–645.
Collins, W. J., D. S. Stevenson, C. E. Johnson, and R. G. Derwent (1997), Tropospheric ozone in a global-scale three-dimensional Lagrangian model and its response to NO_x emission controls, *J. Atmos. Chem.*, *26*, 223–274.
Dentener, F. J., and P. J. Crutzen (1994), A three-dimensional model of the global ammonia cycle, *J. Atmos. Chem.*, *19*, 331–369.
Elvidge, C., K. E. Baugh, E. A. Kihn, H. W. Kroehl, and E. R. Davis (1997), Mapping city lights with nighttime data from the DMSP operational linescan system, *Photogramm. Eng. Remote Sens.*, *63*, 727–734.
Fekete, B. M., C. J. Vörösmarty, and W. Grabs (2002), High-resolution fields of global runoff combining observed river discharge and simulated water balances, *Global Biogeochem. Cycles*, *16*(3), 1042, doi:10.1029/1999GB001254.
Food and Agriculture Organization (FAO) (2001), FAOSTAT 2001 database collections, www.apps.fao.org, Rome.
Food and Agriculture Organization (FAO) (2002), FAOSTAT 2002 database collections, www.apps.fao.org, Rome.
Food and Agriculture Organization (FAO) (2003), FAOSTAT 2003 database collections, www.apps.fao.org, Rome.
Food and Agriculture Organization (FAO)/International Fertilizer Industry Association (IFA)/International Fertilizer Development Center (IFDC) (2003), *Fertilizer Use by Crop*, 5th ed., Rome.
Galloway, J. N., et al. (2004), Nitrogen cycles: Past, present, and future, *Biogeochemistry*, *70*, 153–226.
Garnier, J., P. Servais, G. Billen, M. Akopian, and N. Brion (2001), Lower Seine River and estuary (France) carbon and oxygen budgets during low flow, *Estuaries*, *24*, 964–976.
Garnier, J., G. Billen, E. Hannon, S. Fonbonne, Y. Videnina, and M. Soulie (2002), Modelling the transfer and retention of nutrients in the drainage network of the Danube River, *Estuarine Coastal Shelf Sci.*, *54*, 285–308.

Gerber, P. (2004), FAO-GeoNetwork: Livestock densities, http://www.fao.org/geonetwork/srv/en/main.search, Food and Agric. Org., Rome.

Green, P., C. J. Vörösmarty, M. Meybeck, J. N. Galloway, B. J. Petersen, and E. W. Boyer (2004), Pre-industrial and contemporary fluxes of nitrogen through rivers: A global assessment based on typology, *Biogeochemistry*, *68*, 71–105.

Holland, E. A., F. J. Dentener, B. H. Braswell, and J. M. Sulzman (1999), Contemporary and pre-industrial global reactive nitrogen budgets, *Biogeochemistry*, *46*, 7–43.

Howarth, R. W., et al. (1996), Regional nitrogen budgets and riverine N and P fluxes of the drainages to the North Atlantic Ocean: Natural and human influences, *Biogeochemistry*, *35*, 2235–2240.

IMAGE-team (2001), The IMAGE 2.2 implementation of the SRES scenarios: A comprehensive analysis of emissions, climate change and impacts in the 21st century, *Publ. 481508018*, Natl. Inst. for Public Health and the Environ., Bilthoven.

International Fertilizer Industry Association (IFA) (2002), *IFADATA Statistics From 1973/74-1973 to 2000-2000/01: Production, Imports, Exports and Consumption Statistics for Nitrogen, Phosphate and Potash Fertilizers* [CD-ROM], Int. Fertil. Ind. Assoc., Paris.

International Fertilizer Industry Association (IFA) (2003), *IFADATA Statistics From 1973/74-1973 to 2001-2001/02: Production, Imports, Exports and Consumption Statistics for Nitrogen, Phosphate and Potash Fertilizers* [CD-ROM], Int. Fertil. Ind. Assoc., Paris.

Janssen, P. H. M., and P. S. C. Heuberger (1995), Calibration of process-oriented models, *Ecol. Modell.*, *83*, 55–66.

Krol, M., S. Houweling, B. Bregman, M. Van Den Broek, A. Segers, P. Van Velthoven, W. Peters, F. Dentener, and P. Bergamaschi (2004), The two-way nested global chemistry-transport zoom model TM5: Algorithm and applications, *Atmos. Chem. Phys. Discuss.*, *4*, 3975–4018.

Land Processes Distributed Active Archive Center (2003), Global land cover characteristics database, version 1, http://edcaac.usgs.gov/glcc/glcc.asp, Sioux Falls, S. D.

Lander, C. H., and D. Moffitt (1996), Nutrient use in cropland agriculture (commercial fertilizers and manure): Nitrogen and phosphorous, *RCAIII NRCS Working Pap. 14*, U.S. Dep. of Agric., Washington, D. C.

Lerner, J., E. Matthews, and I. Fung (1988), Methane emission from animals: A global high-resolution database, *Global Biogeochem. Cycles*, *2*, 139–156.

Melillo, J. M., A. D. McGuire, D. W. Kicklighter, B. Moore III, C. J. Vorosmarty, and A. L. Schloss (1993), Global climate change and terrestrial net primary production, *Nature*, *363*, 234–240.

Meyers, T., J. Sickles, R. Dennis, K. Russell, J. Galloway, and T. Church (2001), Atmospheric nitrogen deposition to coastal estuaries and their watersheds, in *Nitrogen Loading in Coastal Water Bodies: An Atmospheric Perspective*, *Coastal and Estuarine Stud*., vol. 57, edited by R. A. Valigura et al., pp. 53–76, AGU, Washington, D. C.

National Research Council (NRC) (1985), *Nutrient Requirements of Sheep*, 6th ed., 112 pp., Comm. on Animal Nutr., Natl. Acad., Washington, D. C.

National Research Council (NRC) (1989), *Nutrient Requirements of Horses*, 5th ed., 112 pp., Comm. on Animal Nutr., Natl. Acad., Washington, D. C.

National Research Council (NRC) (1998), *Nutrient Requirements of Swine*, 10th ed., 210 pp., Subcomm. on Swine Nutr., Natl. Acad., Washington, D. C.

National Research Council (NRC) (2000a), *Nutrient Requirements of Beef Cattle*, 7th ed., 168 pp., Subcomm. on Beef Cattle Nutr., Natl. Acad., Washington, D. C.

National Research Council (NRC) (2000b), *Nutrient Requirements of Dairy Cattle*, 6th ed., 168 pp., Comm. on Animal Nutr., Natl. Acad., Washington, D. C.

Organization for Economic Co-Operation and Development (OECD) (1999), *OECD Environmental Data Compendium 1999*, Paris.

Organization for Economic Co-Operation and Development (OECD) (2001), *Environmental Indicators for Agriculture*, vol. 3, *Methods and Results*, Paris.

Prospero, J. M., K. Barrett, T. Church, F. Dentener, R. A. Duce, J. N. Galloway, H. Levy, J. Moody, and P. Quinn (1996), Atmospheric deposition of nutrients to the North Atlantic Basin, *Biogeochemistry*, *35*, 27–73.

Ramankutty, N., and J. A. Foley (1999), Estimating historical changes in global land cover: Croplands from 1700 to 1992, *Global Biogeochem. Cycles*, *13*, 997–1027.

Resource Observation Systems (EROS) Data Center (2000), Global land cover characterization, http://edcdaac.usgs.gov/, EROS Data Cent. Distrib. Active Arch. Cent., Sioux Falls, S. D.

Seitzinger, S. P., and C. Kroeze (1998), Global distribution of nitrous oxide production and N inputs in freshwater and coastal marine ecosystems, *Global Biogeochem. Cycles*, *12*, 93–113.

Seitzinger, S. P., C. Kroeze, A. F. Bouwman, N. Caraco, F. Dentener, and R. V. Styles (2002), Global patterns of dissolved inorganic and particulate nitrogen inputs to coastal systems: Recent conditions and future projections, *Estuaries*, *25*, 640–655.

Siebert, S. (2005), Global-scale modeling of nitrogen balances at the soil surface, *Frankfurt Hydrol. Pap. 2*, 27 pp., Univ. of Frankfurt/Main, Frankfurt, Germany. (Available at http:/www.geo.uni-frankfurt.de/ipg/ag/dl/publikationen)

Smil, V. (1999), Nitrogen in crop production: An account of global flows, *Global Biogeochem. Cycles*, *13*, 647–662.

Smil, V. (2001), *Enriching the Earth: Fritz Haber, Carl Bosch, and the Transformation of World Food*, 339 pp., MIT Press, Cambridge, Mass.

United Nations Human Settlements Programme (2001) Global urban indicators, vers. 2, http://www.unhabitat.org/programmes/guo, Nairobi.

United States Geological Survey (USGS) (2000), Global Land Cover Characterization (GLCC), classified according to U.S. Geological Survey Land Use/Land Cover System, *Dig. Data Ser. DDS-37*, http://edcwww.cr.usgs.gov/products/landcover/glcc.html, Reston, Va.

Van der Hoek, K. W. (1998), Nitrogen efficiency in global animal production, in *Nitrogen, the Confer-N-s.*, edited by K. W. Van der Hoek et al., pp. 127–132, Elsevier, New York.

Van der Hoek, K. W. (2001), Nitrogen efficiency in agriculture in Europe and India, in *Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection*, edited by J. M. Galloway et al., pp. 148–154, A. A. Balkema, Brookfield, Vt.

Van Drecht, G., A. F. Bouwman, J. M. Knoop, A. H. W. Beusen, and C. R. Meinardi (2003), Global modeling of the fate of nitrogen from point and nonpoint sources in soils, groundwater and surface water, *Global Biogeochem. Cycles*, *17*(4), 1115, doi:10.1029/2003GB002060.

Vitousek, P. M., T. Fahey, D. W. Johnson, and M. J. Swift (1988), Element interactions in forest ecosystems: Succession, allometry and input-output budgets, *Biogeochemistry*, *5*, 7–34.

Vitousek, P. M., P. A. Matson, and K. Van Cleve (1989), Nitrogen availability and nitrification during succession: primary, secondary and old field seres, *Plant Soil*, *115*, 229–239.

Vitousek, P. M., J. D. Aber, R. W. Howarth, G. E. Likens, P. A. Matson, D. W. Schindler, W. H. Schlesinger, and D. G. Tilman (1997), Human alteration of the global nitrogen cycle: Sources and consequences, *Ecol. Appl.*, *7*, 737–750.

Vitousek, P. M., et al. (2002), Towards an ecological understanding of biological nitrogen fixation, *Biogeochemistry*, *57/58*, 1–45.

Vollenweider, R. A., R. Marchetti, and R. Viviani (1992), *Marine Coastal Eutrophication*, 1310 pp., Elsevier, New York.

Vörösmarty, C. J., P. Green, J. Salisbury, and R. B. Lammers (2000), Global water resources: Vulnerability from climate change and population growth, *Science*, *289*, 284–288.

World Resources Institute (WRI) (1998), *World Resources: A Guide to the Global Environment 1998-99*, Washington, D. C.

Zhu, Z. L., and D. L. Chen (2002), Nitrogen fertilizer use in China—Contributions to food production, impacts on the environment and best management strategies, *Nutr. Cycl. Agroecosyst.*, *63*, 117–127.

A. F. Bouwman and G. Van Drecht, Netherlands Environmental Assessment Agency, National Institute for Public Health and the Environment, P.O. Box 303, NL-3720 BA Bilthoven, Netherlands. (lex.bouwman@mnp.nl)

E. W. Boyer, Department of Environmental Science, Policy, and Management, University of California, Berkeley, CA 94720, USA.

P. Green, Complex Systems Research Center, Institute for the Study of Earth, Oceans, and Space, University of New Hampshire, Durham, NH 03824, USA.

S. Siebert, Institute of Physical Geography, University of Frankfurt, P.O. Box 11 19 32, D-60049 Frankfurt/Main, Germany.