Riverine nitrogen export from the continents to the coasts

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[1] We present an overview of riverine nitrogen flux calculations that were prepared for the International Nitrogen Initiative's current global assessment of nitrogen cycles: past, present, and future (Galloway et al., 2004). We quantified anthropogenic and natural inputs of reactive nitrogen (N) to terrestrial landscapes and the associated riverine N fluxes. Anthropogenic inputs include fossil-fuel derived atmospheric deposition, fixation in cultivated croplands, fertilizer use, and the net import in human food and animal feedstuffs. Natural inputs include natural biological N fixation in forests and other noncultivated vegetated lands, and fixation by lightning. We use an empirical model relating total N inputs per landscape area to the total flux of N discharged in rivers based on watershed data from contrasting ecosystems spanning multiple spatial scales. With this approach, we simulate riverine N loads to the coastal zone and to inland waters from the continents. Globally, rivers exported about 59 Tg N yr^{-1}, with 11 Tg N yr^{-1} transported to dry lands and inland receiving waters, and 48 Tg N yr^{-1} transported to the coastal zone. Rates of riverine N loss vary greatly among the continents, reflecting the regional differences in population and the associated anthropogenic N inputs. We compare our estimates to other approaches that have been reported in the literature. Our work provides an understanding of the sources of N to landscapes and the associated N fluxes in rivers, and highlights how anthropogenic activities impact N cycling around the world.

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1. Introduction

[2] Producing food and fuel to meet the demands of the world's population has greatly altered the global nitrogen cycle, increasing amounts of reactive nitrogen (N) in the atmosphere, landscape, and waters [*Smil*, 2001; *Galloway et al.*, 2004]. Because N limits primary production in many waters, the enhanced N supply resulting from human activities has disrupted many ecosystems. This is evidenced by a host of environmental problems facing coastal estuaries and bays which are major water quality concerns worldwide, including intensification of eutrophication, formation of hypoxic zones, and harmful algal blooms [*Vitousek et al.*, 1997; *Burke et al.*, 2000; *Rabalais*, 2002].

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[3] Knowledge of the sources of N to the landscape and the associated transport via rivers to the coastal zone is essential for understanding consequences of N pollution and for developing nutrient management strategies. Nitrogen export varies by many orders of magnitude among the world's rivers [*Caraco and Cole*, 1999]. Various empirical models have been developed to quantify N fluxes that rely on relatively simple descriptors of sources and landscape characteristics. *Alexander et al.* [2002] conducted an intercomparison of several of these models, illustrating that they can provide reasonable quantifications of total N loads in rivers from large regions despite their shortcomings in terms of a physical basis. Six models were applied systematically for 16 large watersheds that span a range of climatic conditions, N sources, and watershed sizes. A regression model put forth by *Howarth et al.* [1996] relating net anthropogenic N inputs to riverine N export was the most accurate and least biased method for predicting N fluxes in these systems among the simple modeling frameworks compared statistically [*Alexander et al.*, 2002].

[4] Several more complex approaches for quantifying N fluxes at the scale of global regions have been presented in the literature recently, providing a window into the uneven patterns of the transport and transformation of N inputs around the globe and implications for riverine N transport. For example, *Van Drecht et al.* [2003], *Green et al.* [2004],

and *Bouwman et al.* [2005] use data on climate, soil properties, landscape characteristics, and river routing to relate biophysical characteristics of regional scale drainage basins to riverine N fluxes from the GEMS/GLORI compendium synthesized by *Meybeck and Ragu* [1995]. Green et al. use aggregate measures of basin temperature and hydraulic residence times of water in soils and river reach networks to characterize transport and transformations of N inputs from landscapes to river basins with a statistical approach. Van Drecht et al. and Bouwman et al. take a more mechanistic approach, aiming to describe the fate and transport of N inputs explicitly (e.g., ammonia emissions, denitrification in soils and waters, leaching and groundwater transport) as they are transported along various hydrologic pathways.

[5] While these approaches provide admirable first approximations at how to characterize disproportionate impacts of N inputs to a heterogeneous landscape, they also point out a vast number of assumptions and challenges associated with characterizing the complex factors controlling N retention and release at this scale [*Green et al.*, 2004; *Van Drecht et al.*, 2003; *Bouwman et al.*, 2005; *Van Drecht et al.*, 2005]. For example, limitations are imposed by: (1) the coarse resolution of spatial databases describing point and nonpoint sources of N (e.g., deposition, fertilizers, human waste, animal manure, crop uptake and fixation); (2) heterogeneous land characteristics and associated assumptions about their hydraulic characteristics (e.g., soils, land cover, geology); (3) characterization of N transformation processes (e.g., assumptions about in-stream retention based on physical flow regime or time-of-travel that do not aim to consider the complexity of chemical and biological conditions that influence microbial activity); and (4) estimates of riverine N loads that are based on sparse observations that are not always representative of mean annual flow conditions, and are synthesized from large numbers of literature sources with differences in measurement methods, sampling frequencies, and length of period covered by the measurements [*Bouwman et al.*, 2005].

[6] In the face of these uncertainties, we assert that simple empirical methods remain useful for exploring broad-scale linkages among N inputs and exports at regional scales. In their original application providing an analysis of N export from large water regions to the North Atlantic Ocean, *Howarth et al.* [1996] found a very strong positive linear relationship between net anthropogenic N inputs per unit land area and N delivery to coastal waters. This method was found to be robust when extended to other spatial and temporal scales [*Alexander et al.*, 2002; *Boyer et al.*, 2002; *Bashkin et al.*, 2002; *Xing and Zhu*, 2002; *Filoso et al.*, 2005]. Given the simple data requirements, we chose this method to estimate global riverine nitrogen transfers to the coastal zone for the International Nitrogen Initiative's current global assessment of nitrogen cycles: past, present, and future [*Galloway et al.*, 2004]. Our goal in this paper is to provide the details of the riverine N export calculations used in that recent global synthesis. Our riverine N exports were cited therein as "Boyer et al. in preparation," referring to this manuscript [*Galloway et al.*, 2004]. To quantify N exported to the coastal zone from the world's rivers, we use a modified version of the model put forth by *Howarth et al.* [1996]. We establish N budgets for each watershed region, tallying total N inputs to each region from anthropogenic and natural sources. We develop a relationship between these inputs and riverine N export using data from watershed studies from contrasting ecosystems spanning multiple spatial scales. Our N budgeting approach is useful, because it allows assessments of the relative importance of the various sources of N within and among regions and provides a systematic method to relate N inputs to riverine responses. Our work provides an understanding of the sources and magnitude of N inputs to terrestrial landscapes and delivery of N via rivers to coastal and inland waters, and highlights how anthropogenic activities impact N cycling.

2. Methods

2.1. Continental Boundaries

[7] To examine N inputs to the continents and associated riverine N export to the coasts, the first step is to characterize the boundaries of interest. The regional units included in this analysis are Asia, Africa, Europe (including the former Soviet Union, FSU), Latin America, North America, and Oceania (including Australia). These units are collections of countries as defined by the *Food and Agriculture Organization* (*FAO*) [2003]. We used spatial data defining the borders of these continental regions from the Environmental Systems Research Institute's Digital Chart of the World (1993, available at http://www.maproom.psu.edu/dcw/). Next, we quantified the portions of each continent that are hydrologically connected to the coastal zone; that is, regions from which flow and solutes can be transported from landscapes to coastal waters. On the basis of digital terrain data, we mapped the boundaries of inland waters, the areas of the landscape that do not drain to the coastal zone due to the fact that their boundaries are confined internally (Figure 1a). We also mapped inactive, dry land areas of the landscape that do not transmit water to the coast due to insufficient surface water runoff (Figure 1b). Inactive areas were determined using a threshold of 3 mm yr^{-1} representing the minimum upstream runoff required to sustain perennial discharge in river channels from a composite global runoff data set [*Vörösmarty et al.*, 2000; *Fekete et al.*, 2002]. Regions that are not composed of inland or inactive water regions are considered to be the active portion of the landscape that is connected hydrologically to the coasts, contributing flow and solutes to the coastal zone.

2.2. Nitrogen Inputs

[8] To quantify N inputs to each continent, we established budgets describing N inputs that arise from various anthropogenic and natural N sources (Table 1), following a modified version of the approach of *Howarth et al.* [1996]. Our goal was to quantify and sum the "new" inputs of reactive N, referring to reactive N that was either newly fixed within a region or newly transported into a region. Anthropogenic N sources include fertilizer, biological N fixation in cultivated cropland, net imports of N in human

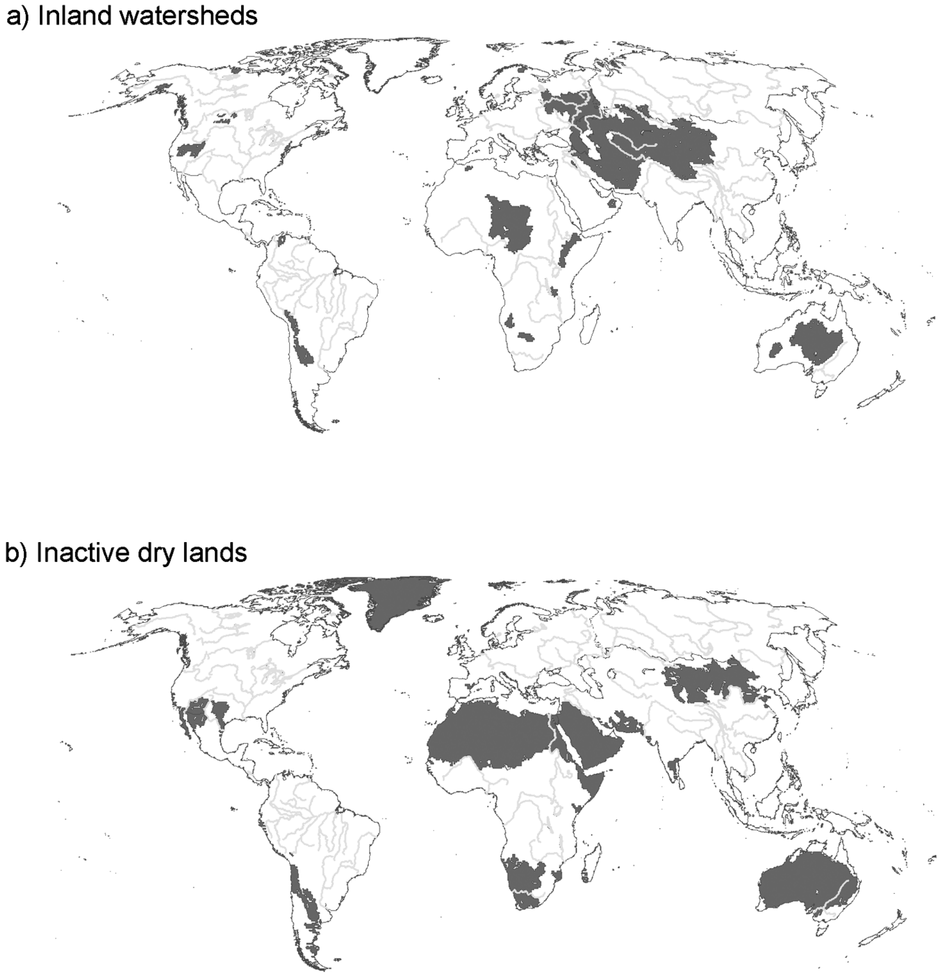

Figure 1. Nitrogen is transported to the coastal zone from rivers draining the landscape. (a) Inland watershed areas (shown in gray) do not drain to the coastal zone due to the fact that their boundaries are confined internally. (b) Inactive areas of the landscape (shown in gray) do not transmit water to the coast due to dry conditions and thus insufficient surface water runoff.

food and animal feedstuffs (where a negative net import term indicates a region that is a net exporter of food and feed), and atmospheric NO_y-N deposition from fossil fuel combustion. The natural sources include biological N fixation in natural (noncultivated) land and N fixation by lightning. Animal waste (manure) and human waste (sewage) are not considered new N inputs because they are recycled within a region; the N in these wastes originated either from N fertilizer, N fixation in agricultural lands, or N imported in food or feeds. Similarly, deposition of ammonium is not considered a new input, as this is largely recycled N volatilized from animal wastes [*Boyer et al.*, 2002]. We assume that the N status of soils is in steady state, with the rate of soil N mineralization equaling the rate of N immobilization over a multi-year period [*Howarth et al.*, 2002; *Galloway et al.*, 2004]. Details of our methods for estimating these individual N input terms from human and natural sources at the global scale (Table 1) have been presented in detail in other studies [*Howarth et al.*, 1996; *Boyer et al.*, 2004; *Galloway et al.*, 2004], and thus are described briefly below. Collectively, the sum of all of the anthropogenic and natural input terms described below are taken to be the total net nitrogen inputs (TNNI) to each region per unit area of the landscape (Table 1). All N budget data are presented in units of mass per time (Tg N yr^{-1}; 1 Tg N = 1 million metric tons N).

2.2.1. Synthetic Fertilizers

[9] Significant quantities of N are deliberately added to the landscape in agricultural regions in the form of synthetic fertilizers, aiming to provide essential nutrients needed to maximize crop production. Globally, the production and application of N fertilizers is the single largest anthropogenic source of reactive N to landscapes [*Boyer et al.*, 2004; *Galloway et al.*, 2004]. To describe the pattern of N fertilizer use, we used country-level estimates of nitrogenous fertilizer consumption from the Food and Agriculture

Table 1. Total Net Nitrogen Inputs, TNNI, Defined as the Sum of Reactive Nitrogen Inputs From Anthropogenic and Natural Sources

Source	Input
Anthropogenic	net atmospheric N deposition
Anthropogenic	nitrogenous mineral fertilizer use
Anthropogenic	net imports of N in food and feed
Anthropogenic	biological N fixation in cultivated agricultural lands
Natural	biological N fixation in forests and other noncultivated vegetated lands
Natural	N fixation by lightning

Organization (FAO) of the United Nations [*FAO*, 2003]. The net N input of synthetic N fertilizer in any region represents the difference between creation of N fertilizers in the regions and the net trade (import or export) of fertilizers between regions. The exchange of N associated with the distribution and use of fertilizers described here does not include the additional N exchanges associated with the trade of crops, though the net exchange of N in crops is included below in the section on food transfers.

2.2.2. Atmospheric Deposition

[10] The major sources of N to the atmosphere are emissions from fossil fuel combustion (e.g., from industries and automobiles) and emissions from agriculture (e.g., from livestock and volatilization of fertilizer and manure). Once emitted, the N molecules can travel long distances in the atmosphere before being deposited on land or water via precipitation (i.e., wet deposition) or through the settling out of particles and gases (i.e., dry deposition). N deposition associated with industrial, automotive, and biogenic N emissions provides significant N input to the landscape at the regional scale. We consider atmospheric N deposition inputs via oxidized forms (NO_y), which comes from anthropogenic burning of fossil fuels [*Howarth et al.*, 1996; *Prospero et al.*, 1996]. We obtained modeled estimates of total (wet + dry) atmospheric deposition of NOy-N associated with fossil fuel burning (industry, power generation, transport) and natural sources (biogenic, lightning) for 1993 from the global chemistry transport model (version TM3) of the University of Utrecht [*Lelieveld and Dentener*, 2000]. This model has been widely used and validated extensively for N species [*Holland et al.*, 1999]. To avoid double accounting of N in our calculation of new, net atmospheric N inputs, we excluded considering N inputs from the volatilization and deposition cycle of both reduced (NH_x) and organic forms of N which are likely to be both emitted and redeposited within our regional watershed boundaries [*Howarth et al.*, 1996].

2.2.3. Food and Feed Transfers

[11] A significant redistribution of N on the landscape occurs due to distribution human food and animal feed. Both humans and animals located within a region of interest consume food and feed, and their nutritional needs are met both through both local agricultural production and importation from other regions. The world's population is unevenly distributed over the landscape, and it is typical to find very dense urban development and high population densities located in lowland areas along rivers and coasts. Food is imported to population centers from agricultural areas, which often are located in fertile uplands and flood plains that support intense crop and animal production. The majority of crops that are grown in agricultural areas of the world are used to feed livestock which, in turn, are consumed by the population in the form of meat, milk, and eggs. Associated with this food production cycle is the vast amount of waste produced by animals (in manure) and by humans (in septic and sewage). Both of these types of waste are very rich in N content, and though some of this waste is transformed naturally in the landscape into atmospheric forms, a significant fraction also is released into surface waters as N. Point sources of waste (e.g., treated human waste from sewage treatment plants) and nonpoint sources of waste (e.g., from leaching of manure) are leading sources of surface water pollution.

[12] We estimated annual net N exports in food and feed for 1995 using import, export, and production data for crops and animals from the FAO agricultural trade databases [*FAO*, 2003] and based on N contents reported in the literature [*Lander and Moffitt*, 1996; *Bouwman and Booij*, 1998]. We assumed that net N import in food and feed is equal to the difference between N demands for human and animal populations in each region, and N produced to satisfy those needs in crop and animal production in each region [*Howarth et al.*, 1996; *Boyer et al.*, 2002]. Thus net import in food and feed = human consumption + animal consumption − crop production for animal consumption − crop production for human consumption − animal production for human consumption. We disaggregated the country scale data to the scale of our regions of interest based on their fraction of area within each watershed. Cases where the balances are negative, with crop and animal production exceeding human and animal demands indicate a net export of N in food and feed. At the continental scale, for example, North America, Latin America, and Oceania are net exporters of N in food, highlighting the importance of world agricultural trade [*Galloway et al.*, 2004].

2.2.4. Fixation in Human-Cultivated Lands

[13] Reactive N is also introduced to the landscape in significant quantities via biological N fixation (BNF) in cultivated land. Many of the crops cultivated in agricultural lands are grown in order to provide a protein-rich (thus N-rich) diet to animals, with a smaller portion going to feed humans directly. Fixation is an anthropogenic source of N to the landscape, owing to the fact that humans are planting vast areas of soybeans, peas, and other crops that host symbiotic nitrogen-fixing bacteria that produce N biologically. To quantify BNF due to human cultivation of crops, we calculated the annual agricultural fixation for 1995 using crop areas and yields reported by the *FAO* [2003]. We

multiplied the area planted in leguminous crop species by the rate of N fixation specific to each crop type, assigning rates recommended by *Smil* [1999, 2001].

2.2.5. Fixation in Natural Vegetation

[14] Biological N fixation associated with tree species and other natural vegetation accounts for nearly 26% of the net anthropogenic N inputs at a global scale [*Boyer et al.*, 2004; *Galloway et al.*, 2004]. To estimate natural BNF inputs to each region, we used modeled estimates presented by *Cleveland et al.* [1999] and modified by Cleveland and Asner based on estimates of plant N requirement simulated with the TerraFlux biophysical-biogeochemical process model to constrain estimates of BNF in vegetation across biomes of the world (C. C. Cleveland and G. P. Asner personal communication, 2002, and *Asner et al.* [2001], as cited by *Boyer et al.* [2004] and *Galloway et al.* [2004]).

2.2.6. Fixation by Lightning

[15] Natural lighting formation is also a natural source of BNF, providing sufficient energy to convert atmospheric N_2 to reactive N [*Vitousek et al.*, 1997]. However, this is a very small source of N in continental world regions. Lightning accounts for only about 2% of the global total net N inputs, and inputs are higher in tropical regions and other regions characterized by high convective thunderstorm activity [*Galloway et al.*, 2004]; lightning accounts for roughly 4% of total net N inputs in Africa and in Latin America. We obtained modeled estimates of total N fixation via lightning for the early 1990s, linked to convection estimates derived from the global chemistry transport model (version TM3) of the University of Utrecht [*Lelieveld and Dentener*, 2000], and based on the parameterization of *Price and Rind* [1992].

2.2.7. Total Net Nitrogen Inputs

[16] Finally, we quantified TNNI to the active and inactive portions of each continent by summing the contributions from each of the anthropogenic and natural N sources (see Table 1). Spatial data from the various data sources described for each of the input terms were aggregated from their associated reporting units (e.g., grid cells or country boundaries) to the scale of the continental regions using GIS software, weighting the fraction of land area that is included within the boundaries of interest.

2.3. Nitrogen Exports

[17] Next we consider relationships between TNNI and riverine N export. In their analysis of the large water regions draining to the North Atlantic Ocean, *Howarth et al.* [1996] put forth an empirical model relating net anthropogenic N inputs per landscape area to the total N export, finding a strong positive linear relationship between the two. Their approach considered new inputs of N to a region that are human controlled, including inputs from fossil-fuel-derived atmospheric deposition, fixation in cultivated croplands, fertilizer use, and the net import in food and feed. Subsequent studies have found that the form of the relationship holds when considering other temperate regions of the world over multiple scales [e.g., *Boyer et al.*, 2002]. Here we have modified the approach to include new inputs of N to a region from natural BNF in addition to the anthropogenic N inputs. Again, the sum of the anthropogenic and natural inputs is defined as the total net nitrogen inputs (TNNI) to each region, per Table 1. We extend the modified model of Howarth et al. to other regions of the world, using data from watershed studies from contrasting ecosystems spanning multiple spatial scales. It remains a challenge to identify basins for which enough long-term monitoring data are available from which to calculate reasonable approximations of N loading in rivers. We scoured the literature for data from watersheds from which there were quality estimates of N inputs at the watershed scale, along with monitoring data on average annual fluxes of total N (in particulate, dissolved, and organic forms) that were based on monitoring data over a range of flow conditions and that represent conditions in the mid-1990s, consistent with the timeframe our N input data at the continental scale. For these watershed-scale N budgets, we use local scale data describing TNNI as described in the publications from which we assembled data (not the coarser continental scale databases described above), relating them to the reported exports of N in rivers. Building on the database for the water regions draining to the North Atlantic Ocean from North America and Europe [*Howarth et al.*, 1996], we added data from watersheds in the northeastern United States [*Boyer et al.*, 2002], Ecuador [*Borbor et al.*, 2005], Brazil [*Filoso et al.*, 2005], Korea [*Bashkin et al.*, 2002] and China [*Xing and Zhu*, 2002]. We establish N budgets for each of the 39 watersheds for which we were able to obtain complete N budget data, tallying TNNI and relating that to riverine N export via linear regression. The resulting equation relating TNNI to riverine N export, which is based on watershed observations from temperate and tropical areas, is subsequently used to predict riverine N export from the continents to the coasts as a function of their N inputs.

3. Results and Discussion

[18] Using the approach described above, we establish total net N inputs (TNNI, see Table 1) to each of the continents, illustrating the unequal distribution of reactive N inputs to the global landscape [*Galloway et al.*, 2004; *Boyer et al.*, 2004]. Natural sources, mostly from biological nitrogen fixation, dominate the N budgets in Africa (79%), Oceania (79%), and Latin America (72%). In contrast, anthropogenic N sources are greater than natural sources in Asia (74%), North America (61%), and Europe/FSU (59%). Globally, fertilizer use is currently the dominant source of new N inputs to the landscape, and is projected to increase significantly in the coming decades, especially in developing regions [*FAO*, 2003; *Wood et al.*, 2004].

[19] To quantify the relationship of TNNI to riverine N export, we extended the model put forth by *Howarth et al.* [1996] to other areas of the world, and established complete input-output budgets for 39 watersheds spanning a range of world regions. There is a direct, linear relationship between TNNI and the total N fluxes in these watersheds which suggests that, on average, 75% of the N inputs are retained in the landscape (that is, stored in soils and vegetation, or lost via volatilization and denitrification), while 25% of the N inputs are exported to rivers (Figure 2; R^2 = 0.61).

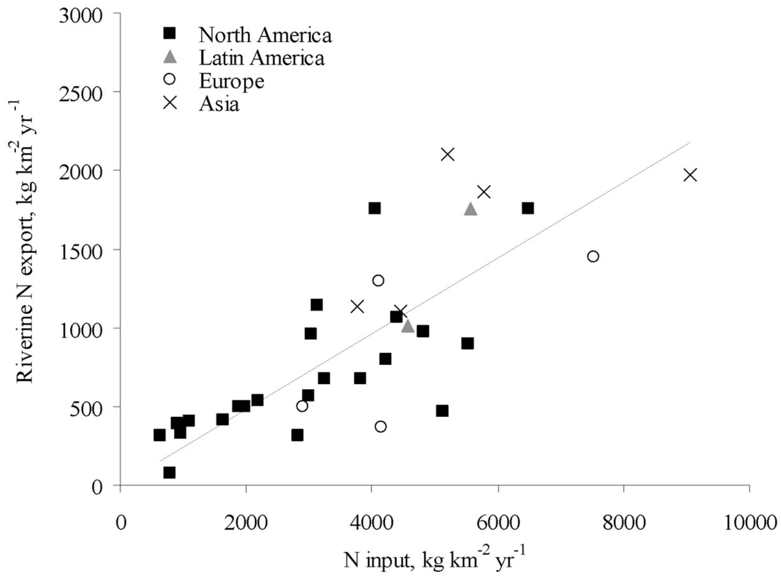

Figure 2. Monitoring data of riverine N fluxes from watersheds spanning multiple scales and biomes of the world (North America, Latin America, Europe, and East Asia) indicate that N inputs are directly related to riverine N exports.

However, there is a lot of variability in the percentage of N inputs exported from the various regions, ranging from about 10–40%. The residuals from the regression line indicate the heterogeneity of the landscape, reflecting the many complex factors that cause variable retention of N inputs. Regardless of how one conceptualizes the transport and transformation of N inputs as they pass through the terrestrial landscape, it is clear from many studies that the greater the N loadings to a region, the greater the potential for riverine N losses [*Howarth et al.*, 1996; *Seitzinger et al.*, 2002; *Van Drecht et al.*, 2003; *Galloway et al.*, 2004; *Green et al.*, 2004; *Goulding*, 2004; *Dumont et al.*, 2005]. A recent multivariate approach put forth by *Howarth et al.* [2005] based on N budgets for 16 catchments in the northeastern USA suggests that a good deal of the unexplained (residual) variance in the relationship between N inputs and N export can be explained by climatic factors (precipitation and discharge). This suggests that wetter areas export a greater percentage of the N inputs than do drier areas. Though we did not utilize a multivariate approach in this analysis, some of the observations in Figure 2 support such an interpretation at first glimpse. For example, the percentage of TNNI exported to rivers is much greater (falling above the regression line) for the Northwest European Coast watershed (which falls above the regression line in Figure 2) than for the Southwest European Coast watershed (which falls below the regression line in Figure 2). The importance of climatic effects on N retention at regional scales is also supported by the aforementioned biophysical approaches that were employed by *Van Drecht et al.* [2003] and *Green et al.* [2004]. Given the fact that the continental boundaries included in this study each span a broad climatic gradient, the simple average provided by the linear regression equation used herein is likely fairly robust.

[20] We use the relationship based on watershed observations from around the world as a first approximation to predict riverine N exports from each of the continents, predicting riverine N export as a function of TNNI. At a global scale, we estimate that rivers exported about 59 Tg N yr^{-1}, with 11 Tg N yr^{-1} transported to inland receiving waters and dry lands and 48 Tg N yr^{-1} transported to coastal waters. Rates of riverine N export vary greatly among the continents, reflecting the regional differences in N inputs (Figure 3). Associated with the need to provide food and energy to its vast population, agricultural practices in Asia are intensive and extensive, and the region receives the highest rates of atmospheric deposition globally [*Galloway et al.*, 2004]. This significantly affects water quality, as riverine N fluxes from Asia to the coastal zone (16.7 Tg N yr^{-1}) and to inland waters (5.1 Tg N yr^{-1}) are the largest in the world. Riverine N fluxes to the coastal zone result primarily from anthropogenic N sources in Europe and the former Soviet Union (8.4 Tg N $yr-^{1}$), and in North America (7.2 Tg N yr^{-1}). Natural sources dominate riverine N fluxes to the coastal zone in Africa (6.6 Tg N yr^{-1}) and Latin America (8.2 Tg N yr^{-1}). Apart from Antarctica, Oceania is the driest continent globally, with deserts or semi-arid landscapes covering most of mainland Australia. Thus riverine fluxes of N in Oceania are small compared to other continents, with more N transported to inland receiving waters/drylands (1.5 Tg N yr^{-1}) than is transported to the coastal zone (0.7 Tg N yr^{-1}) in the mid-1990s. Unlike the United States and Europe which have stabilized rates of population growth, East Asia continues to see very rapid increases in population, agriculture, and industrial activity, and will continue to drive the global riverine N budget.

[21] Understanding the sources and fate of N inputs to watersheds is necessary for mitigating N pollution problems

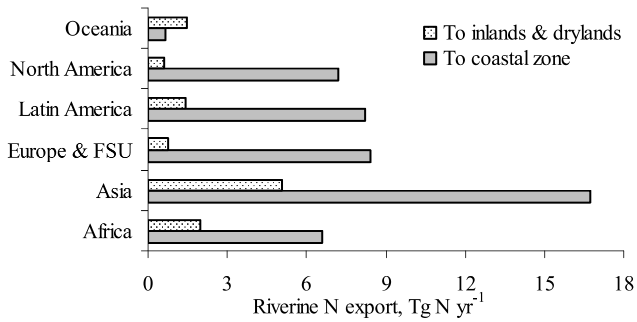

Figure 3. Riverine export of total dissolved nitrogen from continental regions. Some nitrogen transported in rivers is not delivered to the coastal zone but rather to inland receiving waters or to drylands where there is not much throughput of water or solutes (dotted line). Most nitrogen transported in rivers is delivered to the coastal zone (gray).

in coastal and inland waters. Our estimates of N sources, storages, and losses are highly uncertain. Our ability to make these estimates is dependent on the availability a wide variety of statistical and spatial databases, and highlights the need for long-term monitoring. One element of uncertainty comes from the quality of these data themselves and the methods used to scale information to the space and time scales of interest. For example, recent papers discuss challenges in estimating atmospheric N inputs [*Meyers et al.*, 2001] and N loads in rivers [*Brock*, 2001] on the basis of incomplete and uncertain data from point monitoring networks. A recent effort by *Van Drecht et al.* [2005] assesses global databases used in regional scale N models from four different research groups, comparing the magnitude and spatial distribution of individual N input terms and the associated net N surface balance. There is significant variation on the distribution of the input datasets themselves used to characterize N sources in the terrestrial landscape, with largest discrepancies among the datasets compared in data describing the spatial distribution of atmospheric N deposition and biological N fixation. This work also concludes that there is a need for a better understanding of the spatial allocation and description of land use, agricultural production, and agricultural management practices, which vary widely by region [*Van Drecht et al.*, 2005]. Another element of uncertainty comes from the empirical and process models from which we calculate N storage and loss terms. Modeling rates and spatial distribution of nitrogen inputs and fate remains a major challenge, reflecting the complexities of a multitude of different sources, transport pathways, chemical transformations, and parameterizations within the model structures [*National Research Council* (*NRC*), 2000; *Alexander et al.*, 2002]. Further research is needed to better understand the processes controlling N transport in rivers and how to assess these at the scale of large regions.

[22] A number of global datasets have become available recently, and thus there are a number of new approaches that make use of this information to characterize N fluxes at the global scale [*Galloway et al.*, 2004; *Van Drecht et al.*, 2005]. Several different research groups are taking advantage of the availability of these spatial data to quantify riverine N exports at regional scales. The fact that there are commonalities between both the input and output data sets being used by these research groups urges caution in the interpretation of the similarity or differences among results. However, given several independent efforts to consider how best to relate N inputs to riverine N exports it is useful to explore how the results "stack up" against one another. We compare our results to other recent estimates at the global scale for the timeframe of early to mid-1990s. Our approach (this paper, and reported by *Galloway et al.* [2004]) is based on a calibration relating riverine fluxes to net nitrogen inputs and describes total riverine N export, including dissolved, particulate, and organic forms. Globally, we estimate that rivers transport 48 Tg N yr^{-1} to coastal regions, and 59 Tg N yr^{-1} to all waters including the terrestrial inlands and dry lands. Several other recent studies present global riverine loading estimates that are comparable (Figure 4). A total N flux of 40 Tg N yr^{-1} was predicted by *Green et al.* [2004] with an empirical model relating watershed characteristics to N export, which uses export coefficients based on basin temperature and hydraulic residence times in soils, rivers, lakes and reservoirs to transport N loads to river mouths. *Van Drecht et al.* [2001, 2003] and *Bouwman et al.* [2005] use various versions of a conceptual model based on point and nonpoint sources of N coupled with hydro-ecological models describing their fate and transport in the landscape, producing global estimates of total N fluxes of 54 Tg N yr^{-1} [*Van Drecht et al.*, 2001], 54 Tg N yr^{-1} [*Van Drecht et al.*, 2003], and 46 Tg N yr^{-1} [*Bouwman et al.*, 2005]. *MacKenzie et al.* [2002] use the Terrestrial Ocean Atmosphere Ecosystem Model, a biogeochemical model representing couplings among the major element cycles, to estimate a total riverine N flux of 46 Tg N yr^{-1} in dissolved, particulate, and organic forms. Seitzinger and coworkers use an empirical regression approach to quantify N fluxes as a function of anthropogenic inputs associated with population (including sewage, fertilizers, and deposition) and runoff, yielding estimates of 21 Tg N yr^{-1} as dissolved inorganic N [*Seitzinger and Kroeze*, 1998] and 23 Tg N yr^{-1} as particulate N [*Seitzinger et al.*, 2002], with a global total N export of 44 Tg N yr^{-1}. A recent update to this approach by this research group is a set of new empirical models reported by *Seitzinger et al.* [2005], with a modified accounting of N inputs (including, for example, manure and biological N fixation) and variables influencing N retention and loss, yielding a global total N export of 66 Tg N yr^{-1}, contributed from dissolved inorganic (25 Tg N yr^{-1} [*Dumont et al.*, 2005]), particulate (30 Tg N yr^{-1} [*Beusen et al.*, 2005]), and organic (11 Tg N yr^{-1} [*Harrison et al.*, 2005]) forms. Similarly, *Smith et al.* [2003] use an empirical regression approach to quantify N fluxes as a function of population density and runoff, and use a statistical clustering procedure yielding estimates of global riverine N fluxes in dissolved inorganic form ranging between 19 and 23 Tg N yr^{-1}, but do not estimate fluxes in particulate or organic forms. There are advantages and disadvantages of each of the various approaches. In general, the calibration approaches give

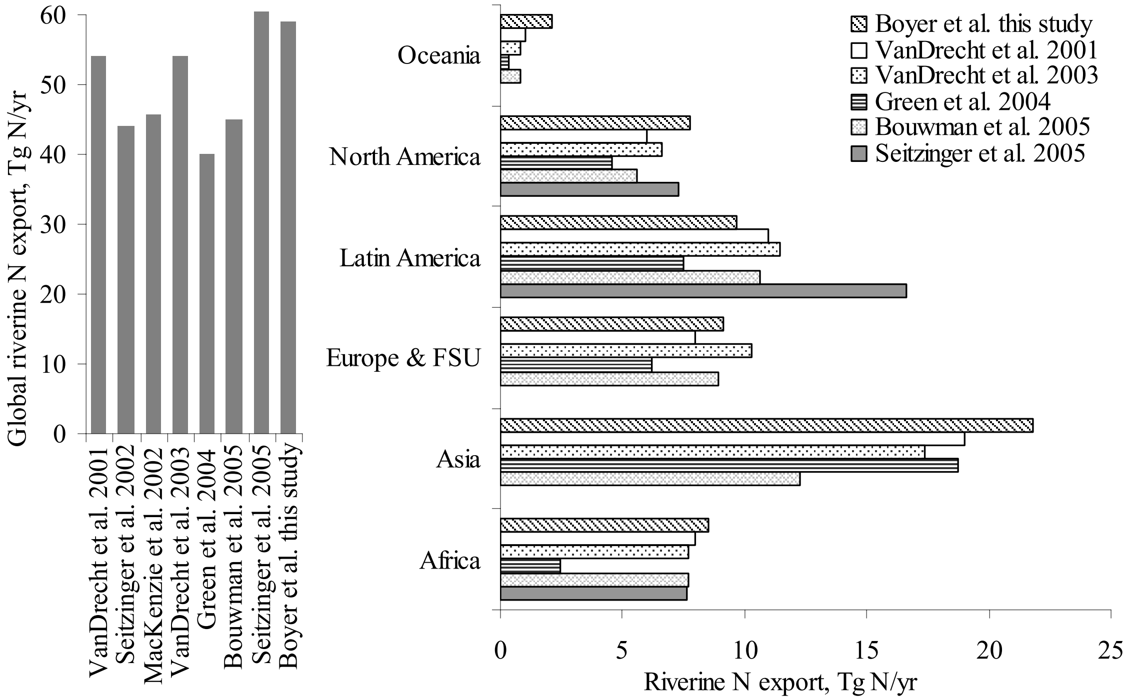

Figure 4. Comparison of estimates of total riverine N export (Tg N yr^{-1}) from studies reported in the literature for the mid-1990s (a) worldwide and (b) by continent.

better predictions for rivers with observations while conceptual models improve our understanding of processes controlling the fluxes. All of the approaches described herein are useful for thinking about past, current, and future scenarios, because in all cases changes in riverine N flux are driven by changes in the magnitude and distribution of the N source inputs. The differences in the estimated riverine N fluxes at regional and global scales highlight large uncertainties stemming from data quality and resolution, scaling issues, and model approaches [*Van Drecht et al.*, 2005].

[23] Notwithstanding the variability in the magnitude of the various riverine N flux estimates at regional scales, there is a generally agreement regarding the unequal distribution of N loading across the continents, reflecting human needs for food and fuel in these world regions. There is a consensus among the scientific community that riverine N fluxes are directly related to N inputs, and that N inputs will continue to increase in the future [*Howarth et al.*, 2002; *Seitzinger et al.*, 2002; *Galloway et al.*, 2004; *Green et al.*, 2004; *Wood et al.*, 2004; *Bouwman et al.*, 2005]. There is also agreement that excess N inputs have adverse environmental consequences [*Howarth et al.*, 1996; *Vitousek et al.*, 1997; *NRC*, 2000; *Townsend et al.*, 2003]. Despite the uncertainties in our riverine N export calculations, our results highlight the need to implement strategies to reduce N loadings to land and water in order to minimize N losses and protect water quality.

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