Global Biogeochemical Cycles

COMMISSIONED MANUSCRIPT
10.1029/2018GB006060

GRAND CHALLENGES
CENTENNIAL COLLECTION

Quantifying Nutrient Budgets for Sustainable Nutrient Management

X. Zhang[1], E. A. Davidson[1], T. Zou[1], L. Lassaletta[2], Z. Quan[1,3], T. Li[4], and W. Zhang[4]

[1]Appalachian Laboratory, University of Maryland Center for Environmental Science, Frostburg, MD, USA, [2]CEIGRAM/Department of Agricultural Production, Universidad Politécnica de Madrid, Madrid, Spain, [3]Institute of Applied Ecology, Chinese Academy of Sciences, Shenyang, China, [4]Key Laboratory of Plant-Soil Interactions, Ministry of Education, Center for Resources, Environment, and Food Security, China Agricultural University, Beijing, China

Key Points:
- Sustainable nutrient management, a major challenge of this century, requires better quantification of nutrient budgets
- This paper reviews major challenges in defining, quantifying, and applying nutrient budgets on multiple system and spatial scales
- The improvement and limitation in nutrient budgets need to be communicated among researchers and stakeholders to enable positive changes

Supporting Information:
- Supporting Information S1
- Table S1

Correspondence to:
X. Zhang,
xin.zhang@umces.edu

Citation:
Zhang, X., Davidson, E. A., Zou, T., Lassaletta, L., Quan, Z., Li, T., & Zhang, W. (2020). Quantifying nutrient budgets for sustainable nutrient management. *Global Biogeochemical Cycles*, *34*, e2018GB006060. https://doi.org/10.1029/2018GB006060

Received 13 AUG 2019
Accepted 9 FEB 2020
Accepted article online 11 FEB 2020

Abstract Nutrients, such as nitrogen and phosphorus, provide vital support for human life, but overloading nutrients to the Earth system leads to environmental concerns, such as water and air pollution on local scales and climate change on the global scale. With an urgent need to feed the world's growing population and the growing concern over nutrient pollution and climate change, sustainable nutrient management has become a major challenge for this century. To address this challenge, the growing body of research on nutrient budgets, namely the nutrient inputs and outputs of a given system, has provided great opportunities for improving scientific knowledge of the complex nutrient cycles in the coupled human and natural systems. This knowledge can help inform stakeholders, such as farmers, consumers, and policy makers, on their decisions related to nutrient management. This paper systematically reviews major challenges, as well as opportunities, in defining, quantifying, and applying nutrient budgets. Nutrient budgets have been defined for various systems with different research or application purposes, but the lack of consistency in the system definition and its budget terms has hindered intercomparison among studies and experience-sharing among researchers and regions. Our review synthesizes existing nutrient budgets under a framework with five systems (i.e., *Soil-Plant* system, *Animal* system, *Animal-Plant-Soil* system, *Agro-Food* system, and *Landscape* system) and four spatial scales (i.e., Plot and Farm, Watershed, National, and Global scales). We define these systems and identify issues of nitrogen and phosphorus budgets within each. Few nutrient budgets have been well balanced at any scale, due to the large uncertainties in the quantification of several major budget terms. The type and level of challenges vary across spatial scales and also differ among nutrients. Improvement in nutrient budgets will rely not only on the technological advancement of scientific observations and models but also on better bookkeeping of human activity data. While some nutrient budget terms may need decades, or even centuries, of research to be well quantified within desirable levels of uncertainties, it is imperative to effectively communicate to interested stakeholders our understanding of nutrient budgets so that scientists and a variety of stakeholders can work together to address the sustainable nutrient management challenge of this century.

Plain Language Summary Managing nutrients, such as nitrogen and phosphorus, is fundamental, yet challenging, for sustainable development. Nutrients are critical for plant and animal growth in agriculture and in nonagricultural ecosystems and are consequently important for food security and climate stability, as well as human health. However, historical and ongoing increases in nutrient inputs to agriculture, while increasing food production, have also contributed to severe environmental problems, ranging from local water and air pollution to global climate change. Quantifying the nutrient inputs and outputs of a farm, watershed, or any other well-defined boundary advances our knowledge of how to maintain a nutrient balance, which is an essential step toward sustainable nutrient management. So far, many research efforts have been devoted to quantifying nutrient budgets and to improving nutrient management for different systems (e.g., farms and food supply chains) and at different spatial scales (e.g., from a single farm to the entire world). However, due to the complex nature of nutrient cycles and incomplete data sets, challenges remain in quantifying and understanding nutrient budgets to inform policies and actions for sustainable nutrient management. With a systematic review of major challenges in defining, quantifying, and applying nutrient budgets, this paper calls for collective efforts by researchers, farmers, watershed managers, consumers, policy makers, and other stakeholders involved in nutrient

management to improve our ability to balance nutrient budgets and to use that understanding to grow abundant food while avoiding pollution.

1. Managing Nutrients in the Anthropocene

Nutrients are fundamental for the survival and growth of life on Earth. To provide a stable and abundant food supply for human society, agriculture has rapidly intensified during the last century, partly owing to our increasing capability to harness nutrients for agricultural production (e.g., the production of nitrogen fertilizer from the Haber-Bosch process and the use of phosphate rock).

The intensive use of nutrients in agricultural production, along with their use in other human activities, has largely distorted the nutrient cycles in the Earth system, especially for the macronutrients, nitrogen (N), and phosphorus (P) (Sutton et al., 2013). For example, the reactive N added to the Earth system by human activities has increased from less than 20 before 1920 to more than 180 Tg N year^{-1} in 2015, surpassing the annual N fixed by natural processes on land or in the oceans (Fowler et al., 2013). Among all anthropogenic N and P inputs, over 90% are from agricultural production (Cordell et al., 2009; Fowler et al., 2013). This intensive use of nutrients for food production has led to severe environmental issues, ranging from local water and air pollution to global climate change. It has been reported that human disturbance to the N and P cycles have crossed the so-called "planetary boundary," suggesting many impacts on the Earth system, including some that may be irreversible (Figure 1, Steffen et al., 2015).

In contrast to the problems of "too much" on a global scale, many regions also suffer from the problems of "too little" availability of nutrients for agriculture (Sutton et al., 2013). Regions, such as Sub-Saharan Africa, still need more macronutrient inputs (e.g., N, P, and sulfur) to reverse declining soil fertility, boost yield, and nourish the population, but fertilizers, high-quality manures, and mulch are often unavailable or unaffordable. In addition to macronutrients, the lack of micronutrients in food, such as minerals like zinc, iodine, and vitamin A, merits more attention.

Consequently, it is imperative to improve nutrient management in order to feed the population with nutritious food while minimizing unintended and undesirable environmental impacts. However, improving nutrient use requires the ability to measure and monitor the fates of nutrients that are cycling within and escaping from ecosystems, but nutrient budgets have been notoriously difficult to quantify. The objective of this review is to provide a historical overview of the nutrient budgeting efforts and identify key challenges in defining, quantifying, and applying nutrient budgets.

2. A Brief History of Nutrient Budgets

You cannot manage what you cannot measure (Deming, 2018). Quantifying nutrient budgets, namely the nutrients entering and leaving a given system, is the first step toward better nutrient management. This approach is based on the principle of conservation of mass, where the difference between the nutrient inputs (Nu_{inputs}) and outputs ($Nu_{outputs}$, including productive outputs $Nu_{prod-outputs}$ and losses Nu_{losses}) of a system changes the nutrient stock (Nu) within the system (Figure 2).

$$Nu = Nu_{inputs} - Nu_{prod\text{-}outputs} - Nu_{losses}$$

Consequently, the efficiency of nutrient use (NuUE) and the nutrient surplus (NuSur) for a system are defined as follows:

$$NuUE = Nu_{prod\text{-}outputs} / Nu_{inputs}$$

$$NuSur = Nu_{inputs} - Nu_{prod_{outputs}} = Nu_{losses} + Nu$$

The origin of nutrient budgeting dates back to midnineteenth century, when European scientists started to quantify nitrogen in rain and drainage waters in field plots (Boussingault, 1841; Lawes, 1882). As N had just been identified as one of the critical nutrients for plant growth, nutrient budgeting started with a focus on soils and crops as a system, targeted at utilizing nitrogen for plant productivity. Since then, the budgeting

10.1029/2018GB006060

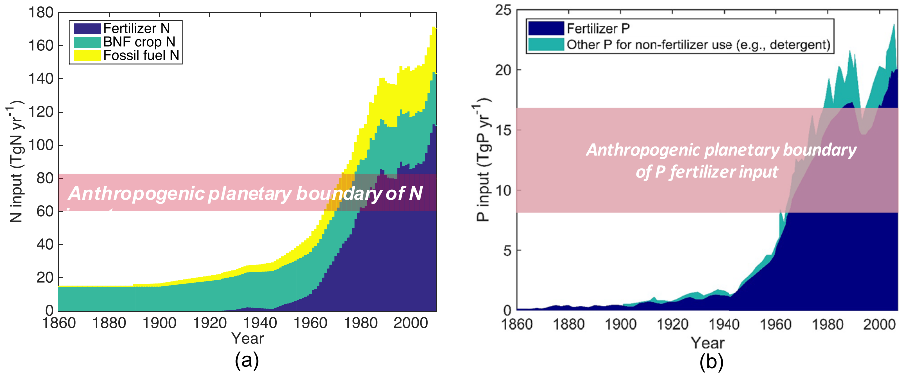

Figure 1. Nitrogen (a) and phosphorus (b) inputs from human activities to Earth system. The data for N and P inputs are from Holland, Lee-Taylor, et al. (2005) and Cordell et al. (2009), respectively. The anthropogenic planetary boundaries of N and P fertilizer input are from Steffen et al. (2015) and Springmann et al. (2018). BNF: Biological Nitrogen Fixation. The green area in Figure 1b denotes the P produced from phosphate rock for nonfertilizer use.

approach, also called soil nutrient balance, has been developed with more input and output items (Allison, 1955) and has been applied to other important nutrients for crop growth, such as phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), and sulfur (S).

Concerned about the depletion of soil nutrients and its consequences on food supply for the United States in early twentieth century, Lipman and Conybeare (1936) took a broader view of nutrient budgets beyond the spatial scale of a few field plots and developed nutrient budgets for all agricultural land for the United States. This development extended the application of nutrient budgeting approach from on-farm management to nutrient management on a national scale. Since the end of World War II, growing amounts of fertilizer have been applied to cropland, especially for N, P, and K, to increase crop yields. These additions have prevented nutrient depletion and, in some cases, have even built up soil nutrient stocks. However, the depletion of soil N and P is still of major concern for food security in many least developed countries, including many Sub-Saharan African countries, and the depletion of soil micronutrients is still prevalent but poorly quantified around the world (Bouwman et al., 2017; Vitousek et al., 2009).

Following the widespread reports of undesirable algal blooms in lakes and estuaries in the midtwentieth century, which has become known as eutrophication caused by excess nutrients (Galloway et al., 2013; Garnier et al., 2010; Glibert et al., 2014), growing concerns about the impacts of fertilizer runoff on water quality and human health have led to development of nutrient budgeting on watershed scales (Howarth et al., 1996; Lowrance et al., 1985; Robertson, 1986; Viets, 1971). Two decades of research have uncovered and solidified the strong linkages between riverine N output and the net anthropogenic nitrogen inputs (NANI) to the watershed (Boyer et al., 2002; Howarth et al., 1996; Jordan & Weller, 1996; Lassaletta et al., 2012; Swaney et al., 2012). The same linkage has also been found for P (Hong et al., 2012). In recent years, the impacts of climate change, especially the changes in precipitation, and the legacy effect of nutrient accumulation on riverine N export have been examined in addition to the anthropogenic N inputs (Powers et al., 2016; Sinha & Michalak, 2016).

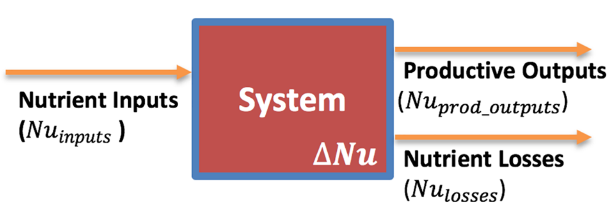

Figure 2. A schematic of nutrient budgets for a system. "Productive outputs" include those nutrients harvested and removed from the system, such as crops, straw, and animal products, which are usually economically valuable products.

Although our emphasis here is on nutrient budgets in which agriculture is a major component, it should be noted that nutrient budgets are also extremely relevant for understanding the biogeochemical processes in nonagricultural systems. The classic works by Bormann and Likens (1979) on the nutrient budgets of forested watersheds of Hubbard Book, New Hampshire, and more recent updates such as Groffman et al. (2018), revealed the impacts of forest management, natural disturbance, secondary succession, and acid deposition in the patterns and processes of forested ecosystems. Similarly, the classic whole-lake nutrient addition experiments initiated by Schindler in 1969 yielded N and P budgets and insights into their roles in freshwater eutrophication processes (Schindler et al., 2008).

The concern over increasing fertilizer use and its impacts on biogeochemical cycles also sparked the quantification of nutrient budgets on a global scale. The SCOPE report edited by Svensson and Söderlund (1976) was among the first to produce global budgets for N, P, and S (Granat et al., 1976; Pierrou, 1976; Söderlund & Svensson, 1976). Meanwhile, in the last quarter of the twentieth century, two global-scale environmental issues, namely stratospheric ozone depletion and climate change, became increasingly recognized, and the contribution of nitrous oxide (N_2O) to these two global issues was identified (Crutzen, 1970; Ehhalt et al., 2001). Consequently, quantifying N budgets, including N_2O, on a global scale has become of interest to both scientists and policy makers (Bouwman, Beusen, et al., 2013; Crutzen et al., 2008; Davidson, 2009; IPCC, 2006; Thompson et al., 2019; Tian et al., 2019; UNEP, 2013). In the recent decade, studies on planetary boundaries further highlighted the importance of quantifying N and P budgets on a global scale, as human impacts on N and P cycles have surpassed the proposed boundaries (Rockstrom et al., 2009; Steffen et al., 2015). After the price spike of phosphate rock in 2008, there was a growing interest in long-term P rock security and the impacts of depleting P rock reserves on food security (Cordell & White, 2014). Although a significant increase in P rock reserves was reported after earlier estimations warning of an approaching peak in phosphorus reserve availability (Cordell et al., 2011; Cordell & White, 2014; USGS, 2019; Van Kauwenbergh, 2010) and researchers have also called into question the basis for a peak P hypothesis (Scholz & Hirth, 2015), there are still concerns on the depletion of rock reserves (especially high grade reserves), the geopolitical consequences of its uneven distribution, and other scarcity issues (Cooper et al., 2011; Cordell & White, 2014; Edixhoven et al., 2014; Khabarov & Obersteiner, 2017).

As the trade-offs between providing nutritious food to a growing population and prevalent nutrient pollution became increasingly acute at the beginning of the 21st century, improvements in nutrient management are not only needed in agricultural production but also needed in a broader system that includes consumers (Sutton et al., 2013). Consequently, nutrient budgeting approaches have extended their application from focusing on production only to considering the whole food system, including consumption and the supply chain. For example, following principles of Life Cycle Assessment, N (or P) footprint analyses quantify N (or P) pollution or resource depletion associated with consuming different food and energy goods by an individual or by an institution such as a university (Erisman et al., 2018; Hoekstra & Wiedmann, 2014; Leach et al., 2012; Leach et al., 2016; Leip et al., 2014; Uwizeye et al., 2016; Xue & Landis, 2010). Conceptual models, such as the Generic Representation of Agro-Food Systems (Billen et al., 2014), have been developed to track N flows among regions and sectors in the food system.

Overall, major motivations for quantifying nutrient budgets include the following:

1. identify and quantify missing nutrient sources or sinks (e.g., Crutzen et al., 2008; Davidson, 2009);
2. evaluate the efficiency of nutrient use in a system, such as the crop and livestock production system or the whole food system (Bai et al., 2018; Bouwman, Goldewijk, et al., 2013; EUNEP, 2015; Lu et al., 2019; MacDonald et al., 2011; Quemada et al., 2020; Zhang, Davidson, et al., 2015);
3. assess the environmental impacts of nutrient losses from a system, such as estimates of N_2O emission from agricultural production based on N inputs or N surplus (Bodirsky et al., 2012; de Vries et al., 2011; Thompson et al., 2019; Tian et al., 2019) or riverine export of N from watersheds based on the NANI methodology (Howarth et al., 2012); and
4. inform stakeholders, such as farmers and policy makers, on the performance of nutrient management and identify how the productivity of the nutrient use can be improved and nutrient pollution can be reduced (Davidson et al., 2016; McLellan et al., 2018).

3. Challenges in Defining Nutrient Budget Systems

Given strong motivations from multiple stakeholders involved in nutrient management, nutrient budgets have been quantified on various spatial and system scales, ranging from a single plot to the whole globe, and from a *Soil-Plant* system to the *Landscape* system (Figure 3). The flourishing application of nutrient budgets has significantly advanced our understanding of nutrient cycles, especially for N and P, but has also been accompanied with confusion, even misunderstanding, over nutrient budgets (i.e., inputs and outputs of a system) and the assessment of the system efficiency, leading to major challenges in comparing nutrient budget results among studies and across scales.

10.1029/2018GB006060

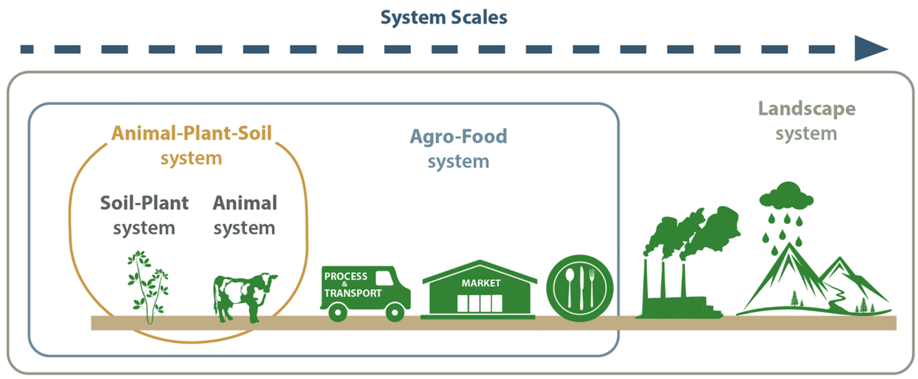

Figure 3. An illustration of the system scales used for quantifying nutrient budgets. The definition of the system scales is developed from Li et al. (2019). The system scales define the key elements that will be examined within a physical boundary (i.e., a spatial scale). For example, in a watershed, the assessment of the nutrient budget can focus on the crop production only (i.e., a type of Soil-Plant system), include the production of livestock (i.e., Animal-Plant-Soil System), or include the whole food supply chain (i.e., Agro-Food System), or all the other human activities that induce N inputs to the landscape, such as fossil fuel burning (i.e., Landscape System).

Nutrient budgets have been quantified for mainly five systems, including *Soil-Plant* system, *Animal* system, *Animal-Plant-Soil* system, *Agro-Food* system, and *Landscape* system (Figure 3 and Table 1). Each of these systems could be applied on most of the four spatial scales, namely Plot and Farm, Watershed, National, and Global scales. Those spatial scales define the physical boundaries for the nutrient budget accounting. Both Watershed and National scales can be considered as regional scales, but one is defined by physical characteristics of the landscape, and the other is defined by political boundaries. Given the goals of the nutrient budget study, a time scale also needs to be selected, ranging from a single growing season to several years or decades (Meisinger et al., 2008).

3.1. *Soil-Plant* System

The nutrient budget of a *Soil-Plant* system, also called soil surface budget (Oenema et al., 2003) or nutrient balance (McLellan et al., 2018), considers the soil and the plants growing in it as an integrated system and accounts for the inputs and outputs of the integrated system. For example, the major N inputs for the system include synthetic fertilizer, atmospheric deposition, biological fixation, and manure. The major outputs include harvested crop products (Figure 4a), which, in addition to grains and other commercial crops, could also include crop residues removed for fuel, feed or other uses off site. In this system, the dynamic nutrient exchanges within soil and between the soil and the plants are often considered as internal processes and are not accounted for in the input/output budget.

The *Soil-Plant* system approach has been increasingly adopted for assessing the efficiency of crop nutrient management, but two challenges associated with this system need to be noted: (1) the efficiency assessment is based on the assumption that the system has achieved a steady state (Nu is 0 or negligible compared to $Nu_{prod\ -\ outputs}$, see Figure 2), but long-term observations are often required to ensure the quasi steady state, especially for cropping systems with multiyear rotation practices, perennial crops, or large nutrient reserves in soil; (2) whether to include crop residue as part of the productive output of the system can lead to significant difference in NuUE assessment. For example, the N use efficiency (NUE) for the *Soil-Plant* system can increase from 43% to over 58% (depending on the harvest index) if crop residue is included as one of the productive outputs. However, the consideration of crop residue should depend on whether the residue is harvested (i.e., removed from the site) and how it is used, for which information is often limited.

Mineralization of soil organic matter provides inorganic N and P available for crop uptake, but it should not be considered as a new input to the *Soil-Plant* system defined here. Rather, it is accounted for as part of the Nu term in Figure 2 if mineralization exceeds immobilization and if the mineralized N or P is taken up by the crop and removed in the harvest. Internal cycling of soil nutrients through mineralization, immobilization, and turnover (MIT) of soil organic matter has been studied extensively for decades, as it is essential for estimating the crop demand that can be met by MIT of organically bound nutrients derived from previous years'

Table 1
Examples of Studies at Various Spatial and System Scales

Spatial\systems scales	Soil-Plant system	Animal system	Animal-Plant-Soil system	Agro-Food system	Landscape system
Plot and Farm scale	Several cases for cropping systems based on fertilization trials (EUNEP, 2015)	Commercial broilers; Patterson et al. (1998)	The cases for mixed crop-livestock farming systems in EUNEP (2015); Powell and Rotz (2015)		
Watershed scale	Chen et al. (2017); Liu et al. (2008)	Poultry layer, broiler, and turkey operations in an aquifer (Zebarth et al., 1998)	The combination of animal production and crop production systems in Ma et al. (2012)	The combination of animal and crop production and food processing in Ma et al. (2012)	Howarth et al. (1996); Hong et al. (2012)
National scale	Bouwman et al. (2017); Zhang, Davidson, et al. (2015)	Pig and poultry systems in Netherland (Oenema, 2006)	The farm N budget for EU countries in Leip et al. (2011)	Billen et al. (2014); Ma et al. (2010); Leach et al. (2016); Le Noë et al. (2017);	Han et al. (2014)
Global scale	Bouwman, Goldewijk, et al. (2013); Zhang, Davidson, et al. (2015)	The pig production system in the whole pork supply chains (Uwizeye et al., 2019)	The combination of feed production and pig production systems in Uwizeye et al. (2019)	Billen et al. (2014); Bodirsky et al. (2014)	Smith et al. (2012)

Note. Gray shaded cells note the categories with existing studies. While it is not an exhaustive list of references, it provides good coverage of the types of systems that have been studied for nutrient budgets. The cases or systems are specified in the table when more than one case or system was examined in a study.

inputs of crop residues to the soil (Lin et al., 2016; Meisinger et al., 2008; Norton & Schimel, 2011). In the context of a multicropping system, MIT imbalance under one crop can have important effects on nutrient availability for the next crop of the cropping system cycle. For example, biological N fixation inputs under soy that are retained as crop residues and incorporated into soil organic matter become important sources of N that partially meet the demand of the following corn crop. The net input/output balance is most relevant for the entire crop rotation period, whereas the internal cycling is highly relevant for varying balances of inorganic, organic, and total nutrient pools within the rotation period. As crop rotations become more complex than simple corn-soy rotations, the internal cycling and MIT estimates become more important for understanding nutrient dynamics within each phase of the cropping cycle.

3.2. *Animal* System and *Animal-Plant-Soil* System

The *Animal* system concerns the animal production only and considers feed production (e.g., from pasture or crop) as a separate system providing nutrient inputs to the *Animal* system; while the *Animal-Plant-Soil* system considers animal production integrated with the crop and/or pasture production; consequently, the locally produced feed and the manure recycled to cropland are both internal processes in this system. Examples of major inputs and outputs of the two systems on a global scale are provided in Figure 4. The example shows that the NUE for the *Animal-Plant-Soil* system is 15%, in comparison to 8% for *Animal* system alone.

Extending the system boundary beyond the animal production facility or farm, the *Animal-Plant-Soil* system enables the consideration of nutrient losses occurring during the production of feed and other inputs for the livestock production. However, where to draw the spatial boundary for accounting the "imported" feed and "exported" manure affects the nutrient budget assessment and consequently challenges the intercomparisons. For example, accounting for the N inputs of imported feed increases the N inputs of the system and consequently reduces the calculated NUE. In contrast, accounting for the manure moved out of the spatial boundary (e.g., sold to another farm) as part of the productive output will increase the system outputs for the farm where it was produced and increase the calculated NUE, even though some portion of the manure N will eventually be lost in the export process (Bai et al., 2014, 2018; Godinot et al., 2015). As a result, the calculated efficiency of a system can vary depending upon how factors like feed imports or manure export are treated in the accounting system, and how recycling within a spatial boundary is considered (Quemada et al., 2020).

3.3. *Agro-Food* System

The *Agro-Food* system extends the system boundary from production to the whole food supply chain and traces nutrient use from agricultural production to marketplaces and to consumers. Many of the nutrient budget assessments for this system investigate not only the inputs and outputs of the system but also the interactions and nutrient flows between its subsystems, such as the *Soil-Plant* system and the *Animal* system. The assessment of nutrient budgets for this system is challenging for two major reasons: (1) The nutrient flows through the food supply chains (e.g., processing and retailing) are very complex, and information is often scarce (Ma et al., 2010). For example, the processing of soybean results in

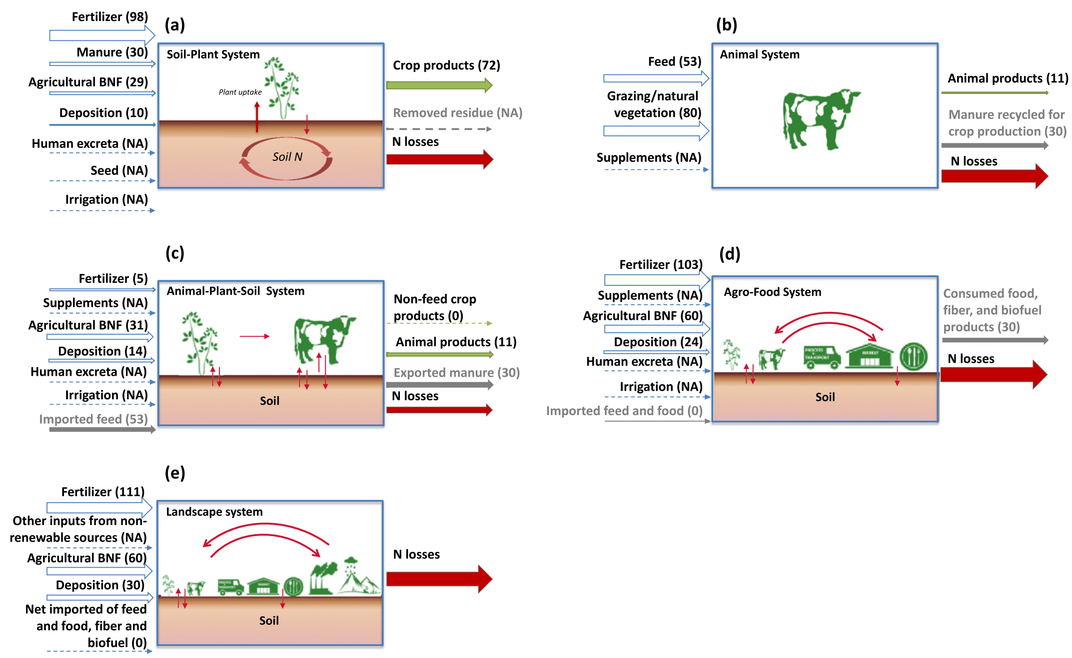

Figure 4. Examples of major inputs and productive outputs of the five systems. The numbers in the parentheses are the estimate of each budget term for N on the global scale with the unit of Tg N year^{-1}. “NA” indicates that the estimate is not available for the budget term in this example. The estimates are based on Billen et al. (2014), with minor modifications as fully described in Table S3.

multiple co-products or by-products, such as oil and soybean meal (Dourado et al., 2011), which can be sold to customers or be recycled within the *Agro-Food* system. Most studies estimate the productive outputs of consumed food based on household surveys or daily-intake guidelines (Leach et al., 2012; Ma et al., 2010), instead of actually following the nutrient flows from the upstream source and through the supply chain. (2) With the increasingly connected interregional and international markets, the food system for a region or a country can rarely stay independent; consequently, it is challenging to draw the geographic system boundary to consider trade and recycling components across spatial boundaries while keeping the tasks of quantifying nutrient budgets manageable (Lassaletta, Billen, Grizzetti, Garnier, et al., 2014; Oita et al., 2016).

To address those challenges, two types of approaches have been taken: (1) the first approach focuses on the nutrient flows involved in producing one or a selection of food products, including the resulting nutrient inputs outside of the defined spatial boundary (e.g., the N and P fertilizer use for producing imported feed, also known as virtual N and P; Erisman et al., 2018; Huang et al., 2019; Leach et al., 2012; Mu et al., 2016; Uwizeye et al., 2019); and (2) the second approach focuses on the function of the food system and its environmental impacts within the spatial boundary; it accepts imported food or feed as nutrient inputs equivalent to others for the *Animal-Plant-Soil system* and ignores the external environmental impacts of producing food and feed imported from outside of the spatial boundary (Ma et al., 2010). This approach shares similar challenges described for the *Animal-Plant-Soil* system above. The two different approaches may result in different assessments for the N budget and the system efficiency, when the defined country/region imports or exports a large amount of feed or food. Taking China as an example, the N inputs and NUE of the *Agro-Food* system are 68.5 Tg and 6%, respectively, based on the first approach, compared to 48.8 Tg and 9% based on the second approach (based on data from Ma et al., 2010; see Table S1 for details).

The first approach, also called a N footprint for the consumed products, has been promoted to guide consumer's dietary choices. In contrast, the spatially defined second approach could be used for adjusting production and trade portfolios for a given region to meet local food demand and reduce regional nutrient pollution. However, few studies have examined the connections between two approaches and their implications in guiding the decision making by individuals and policy makers. For example, the calculated N footprint based on the first approach, which includes the virtual N inputs used outside the region to produce the imported product, may encourage more locally produced food where N inputs might be better managed, but it would, nevertheless, add an environmental burden for the region to support more direct N inputs and their environmental consequences.

3.4. *Landscape* System

The *Landscape* system considers human activities and the terrestrial ecosystem as an integrated system and assesses the "new" nutrient introduced by human activities, including agricultural and industrial production, into the system via synthetic fertilizer application, atmospheric deposition, crop N fixation, and net import of livestock feed and human food (i.e., the difference between imports and exports; Howarth et al., 1996). Unless imported from another landscape region, nutrients in manure and sewage produced within the landscape as well as consumption of locally produced food are considered as internal flows of the system that are converted from the "new" nutrients and consequently are not included in the inputs of the system. The application of this system is mostly known for the NANI or net anthropogenic phosphorus input (NAPI) approaches, a positive relationship between NANI (or NAPI) and riverine nutrient export has been reasonably well established across watersheds (Hong et al., 2012; Swaney et al., 2012).

While the riverine nutrient output has been extensively examined, internal flows with positive or negative effects are less investigated, and the productive outputs of the system are generally not considered. Instead of using only productive nutrient outputs, social and economic products, such as population size or GDP, may be considered as the productive outcomes of the *Landscape* system for assessing how efficiently "new" nutrients are used for supporting human activities for a given region.

4. Challenges in Quantifying Nutrient Budgets

Regardless of the definition of the systems to which nutrient budgets are applied, they are usually difficult to balance, often leading to significant "missing" nutrients (Bouwman, Beusen, et al., 2013). This could be caused by the large uncertainties of many budget terms or potentially some nutrient flows have not been identified or accounted for, such as soil mineralization-sequestration, ammonia volatilization, nitrate leaching, and denitrification. The uncertainties for a nutrient budget term vary across spatial scales because available measurement approaches and data quality vary. Here we review the quantification challenges of eight major nutrient budget terms on four spatial scales, namely Plot and Farm, Watershed, National, and Global scales, and two quantitative examples are presented in Table S2 in the supporting information.

4.1. Nutrient Inputs From Synthetic Fertilizer

Synthetic fertilizers are the major N and P inputs for *Soil-Plant*, *Agro-Food*, and *Landscape* systems examined at the global scale, accounting for over 50% of the total inputs (Figure 4). It also dominates the nutrient inputs in many nutrient budget studies applied to various systems and at various spatial scales. For example, synthetic fertilizer accounts for about 80% of N inputs for maize production in northeast China. Consequently, a potential 10% uncertainty in the synthetic N fertilizer input will result in about 20% uncertainty in the N surplus, which is equivalent to the sum of reactive N losses in the forms of NH_3, N_2O, and leaching and runoff (about 21 kg N ha^{-1}) (Zhang, Ju, et al., 2019). Although 10% seems like a relatively small uncertainty for a budget term, the size of the fertilizer input is often so large that this 10% uncertainty dwarfs other budget uncertainties, even where they are a large percentage of the central estimate. Accurately quantifying the nutrient inputs from synthetic fertilizer is critical for balancing the nutrient budget. In contrast, the uncertainty of a nitrous oxide (N_2O) emission estimate may be ±50% or more due to large spatiotemporal variation of fluxes, but the flux is usually <10 kg N ha^{-1} $year^{-1}$, so this uncertainty term is not large relative to total inputs and outputs at the farm scale, despite being important for global N_2O budgets.

Information on fertilizer inputs is often available at two spatial scales, namely farm and national scales. Farmers (or scientists conducting field experiments) usually have good knowledge of the amount of fertilizer applied in their fields, since it is one of the major costs for their production. On the national scale, countries have been requested to report their fertilizer use for agricultural production, for which they often aggregate their census data on fertilizer use or sales from subnational scales. These data have been compiled and made available by FAOSTAT since 1961 (FAOSTAT, 2019). The International Fertilizer Industry Association (IFA) also reports fertilizer production and consumption data on a national scale based on surveys with fertilizer companies (Heffer, 2013; Heffer et al., 2017). While most countries have reported total synthetic N fertilizer consumption from the two data sources in approximate agreement, a few countries (e.g., China) show

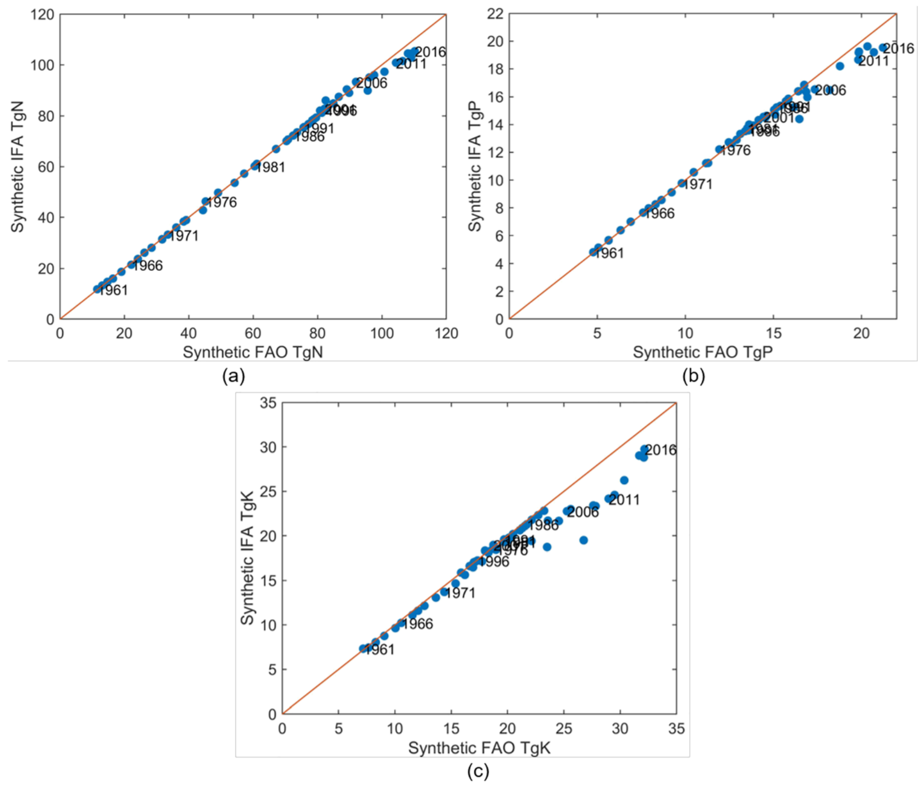

Figure 5. Comparing global total synthetic N (a), P (b), and K (c) consumption reported by FAO (FAOSTAT, 2019) and IFA (IFA, 2019) from 1961 to 2016.

up to 30% underestimation by the Food and Agriculture Organization of the United Nations (FAO) data source, leading to different global fertilizer consumption estimates from the two data sources in recent decades (Figure 5). Additionally, if the N budget assessment focuses on cropland only, then the fraction of synthetic N application used for cropland versus pastures and other noncroplands may become important in determining the N inputs (Lassaletta, Billen, Grizzetti, Anglade, et al., 2014; Xu et al., 2019). This fraction ranges from 12% to 100% among countries (Lassaletta, Billen, Grizzetti, Anglade, et al., 2014) and is likely to decrease when a country becomes more developed (e.g., more fertilizer used for turf-grass) or has more pasture and agro-forestry.

In addition to some disagreement on the national scale from IFA and FAO sources, major gaps in estimating synthetic fertilizer inputs exist between the farm and national scales. Accurate spatially explicit fertilizer input data are urgently needed for nutrient budgeting on subnational or watershed scales, as well as biogeochemical modeling (Lu & Tian, 2017). To arrive at the spatially explicit fertilizer input data, one can upscale the farm-level survey or downscale the national or subnational scale statistics. However, the upscaling method requires farmers to report their fertilization rates, which concerns many farmers due to possible regulation (Osmond et al., 2015; Perez, 2015). The method may also suffer from sampling biases. On the other hand, the downscaling method relies on the distribution of cropland area and assumptions made for fertilizer distribution, both introducing biases in the estimation. For example, Lu and Tian (2017) used the HYDE 3.2 global cropland distribution map to estimate the distribution of N fertilizer inputs globally. Taking India as an example, however, the cropland area from HYDE 3.2 product is over 15% higher than a cropland distribution map from Tian et al. (2014) and about 10% lower than a new Landsat-Based cropland map (GFSAD, 2019; Oliphant et al., 2019). Some downscaling was carried out based on the assumption that fertilizers were applied in the state (or a subnational political unit) where they were sold or adjacent states (Fixen et al., 2012), while others assume a constant fertilizer application rate for all crops (Nishina et al., 2017) or all states (Houlton et al., 2019; Zhang, Davidson, et al., 2015). Only a few efforts have downscaled the fertilizer application rate considering the heterogeneity among regions and crop types (Cao et al., 2018).

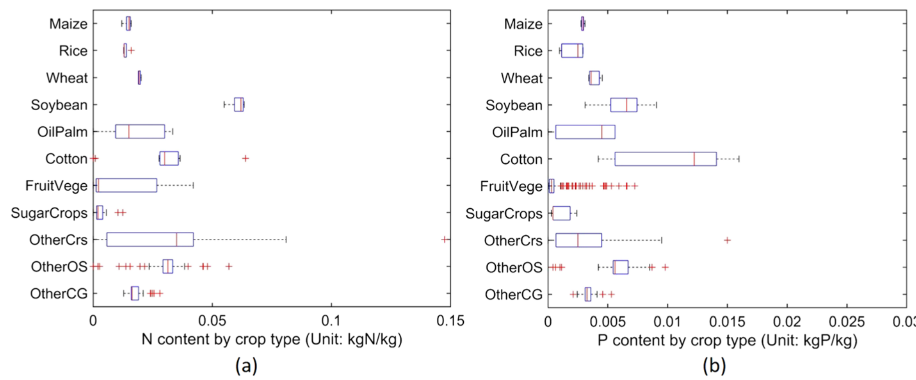

Figure 6. N (a) and P (b) contents of 11 crop or crop groups. N contents are from five data sources (Bodirsky et al., 2012; Bouwman et al., 2017; Feedipedia, 2012; IPNI, 2014; Lassaletta, Billen, Grizzetti, Garnier, et al., 2014), and P contents are from seven data sources (AUSNUT, 2013; Bouwman et al., 2017; FAO, 2006; Feedipedia, 2012; Gourley et al., 2010; IPNI, 2014; USDA, 2013).

The gaps between the upscaling and downscaling approaches demonstrate one of the major challenges in balancing the nutrient budget on watershed scales and other subnational scales (Table S4).

4.2. Nutrients in Crop or Animal Products

Nutrients contained in crop or animal products are essential outputs from most systems examined, are determining factors for NuUE of the system, and are important inputs for several systems (e.g., *Animal* system and *Agro-Food* system) as the feed or imported products (Figure 4).

While the nutrient contents in crop and animal products could be sampled and directly measured in studies on a farm scale, their quantification on broader spatial scales usually relies on the assumed concentrations of nutrients (kg nutrient per kg product) and the recorded product quantity converted to mass (Billen et al., 2014; Lun et al., 2018). However, those nutrient concentrations may vary among regions or even within the same region associated with different varieties and management practices (Guardia et al., 2018). Taking N and P contents in crop products as an example, Figure 6 demonstrates differences of nutrient contents for a crop type or within a crop group using several commonly used data sources (AUSNUT, 2013; Bouwman et al., 2017; FAO, 2006; Feedipedia, 2012; Gourley et al., 2010; IPNI, 2014; USDA, 2013). In addition to the spatial variation, the nutrient contents also change along time as yield changes or cultivars shift. Ciampitti and Vyn (2012) suggested that the N content in maize has decreased from 0.0133 kg N kg^{-1} grain in the period of 1940–1990 to 0.0120 kg N kg^{-1} grain in the period of 1991–2011 on average. This 10% decrease in N contents suggests that N surplus reduction in the United States estimated based on a constant N content (Zhang, Davidson, et al., 2015) may be overestimated by about 15 kg N ha^{-1}. On the other hand, Tenorio et al. (2019) estimated that yield differences explained much more variability in N surplus than variation in N concentration of harvested grain. Unfortunately, few records are available to track this critical change for most crops.

Depending on the data sources, the product quantity estimated for the same spatial boundary (e.g., a nation) may not be consistent among studies. For example, maize yield in China is estimated as 5.5–8.9 tons ha^{-1} based on intensive farm surveys but is reported as 4.7–5.8 tons ha^{-1} according to the government census (Table S4; NBSC, 2019; Zhang, Ju, et al., 2019). Using data from those different sources may help with assessing and constraining the uncertainties associated with this nutrient budget term.

4.3. Nutrients in Manure

Manure excreted during animal production is one of the major pathways by which nutrients are lost to nonagricultural soils or water, causing pollution (Cordell & White, 2014). Part of the manure is recycled to the *Soil-Plant* system, accounting for about 18% and 28% of the global N and P inputs to cropland, respectively (Figure 4). Nutrients in manure could be quantified by direct laboratory analyses of manure or calculated based on animal stocks and documented nutrient excretion rates (kg nutrient head^{-1} year^{-1}; Billen et al., 2014; Bouwman, 1997; Bouwman et al., 2017; Hou et al., 2016; Lassaletta et al., 2019). To measure the

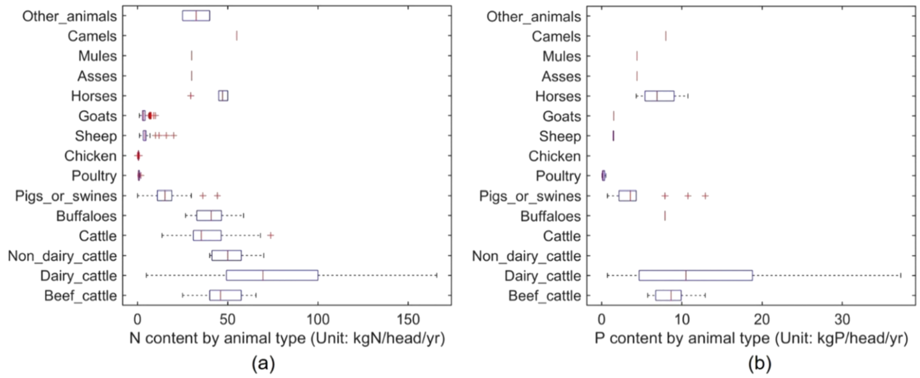

Figure 7. N (a) and P (b) contents in the manure of 15 common animal groups. N contents are from five data sources (Bouwman et al., 2017; FAO, 2018a; Lorimor et al., 2004; Mosier et al., 1998; Van der Hoek, 1998), and P contents are from two data sources (Bouwman et al., 2017; Lorimor et al., 2004). The small variations among data sources for several animal types are likely caused by lack of measurement.

nutrient excretion rate in a lab, MidWest Plan Service provides detailed information on laboratory selection, sampling, and testing for N, P, and K (Lorimor et al., 2004). N excretion rate can also be estimated using the difference between the animal's total N intake and total N needed for growth and milk production (Dong et al., 2006) or assumed to be proportional to the weights of slaughtered animals (Sheldrick et al., 2003; Lassaletta, Billen, Grizzetti, Anglade, et al., 2014).

Similar to nutrients in crops and animal products, quantifying nutrients in manure depends on knowledge of nutrient concentrations and manure mass, which vary by factors such as year, region, weather, animal type, diet, animal age, treatment, storage, and calculation method (Figure 7). However, it is usually assumed that nutrient contents do not change over time and constant nutrient excretion factors per animal are applied at the large region scale (Van der Hoek, 1998). Those assumptions and parameters used for the estimation lead to uncertainties in the quantification. For example, the default N excretion rates provided by Intergovernmental Panel on Climate Change have uncertainty about ±50% (Dong et al., 2006).

Recent efforts have been devoted to developing country- and livestock-specific manure N parameters (e.g., the development of the Global Livestock Environmental Assessment Model; FAO, 2018a, MacLeod et al., 2017). Regularly measuring and reporting manure nutrient concentrations, excretion rates, and the fate of manure by animal type and region will help to improve the quantification of nutrient budgets in the form of manure.

4.4. Atmospheric Deposition

Atmospheric deposition is a relatively small source of N inputs for most agronomic systems examined by the nutrient budget approach, accounting for 6–14% of the total inputs for agricultural landscape globally (Figure 4), but could be important for the N budget of *Landscape* systems in regions with little agricultural activity. For example, N deposition accounts for about 35% of net anthropogenic N inputs to watersheds in the northeast region of the United States. (Hong et al., 2013). Inputs of P through atmospheric deposition are usually minimal and are often neglected in budgets (Mogollón et al., 2018)

The quantification of atmospheric N deposition only accounted for N in rainfall in the early twentieth century (Lipman & Conybeare, 1936) and has evolved to include both wet (e.g., NO_3^- or NH_4^+ in rain or snow) and dry (e.g., reactive N gas or particles) deposition in recent decades (Fenn et al., 2008; Hember, 2018; Schwede & Lear, 2014; Yu et al., 2019). The wet deposition is often determined by the amount of precipitation and the N concentration in precipitation, which can be directly measured in the field. The measurement techniques have been long established, but Erisman et al. (2005) found that the widely used bulk sampler reported higher deposition flux than wet-only sampler by up to 40%. In addition to the influence of dry deposition, the uncertainties in the measurement could be caused by factors such as the placement of samplers and sample handling and analysis. On a spatial scale larger than an experimental site, the quantification of wet deposition requires the interpolation and/or modeling of existing observations at discrete locations, where the availability of observations and the selection of interpolation methods (e.g., kriging) affect the uncertainties in the estimated deposition. Models such as the Community Multiscale Air Quality have been used to simulate atmospheric transport, chemistry, aerosol physics, and deposition at a 12-km horizontal resolution as inputs to N budgets (Pinder et al., 2012). Countries in North America, Europe, and East Asia have established national or regional monitoring networks for deposition (Hember, 2018; Xu et al., 2015), but the observation is still fairly scarce in regions such as Africa and South America.

The quantification of dry N deposition is even more challenging and uncertain than the wet deposition, even though its quantity is comparable to wet deposition (Schwede & Lear, 2014; Xu et al., 2015; Yu et al., 2019). Dry deposition can be directly measured with flux towers or chambers for spatial scales of a plot to a farm, but the measurement is limited by (1) available sensors that can provide robust, high-frequency, and high-accuracy measurement of gas concentrations (Erisman et al., 2005; Zhang, Lee, et al., 2015); and (2) logistical resources needed for maintaining a long-term observation. So far, systematic monitoring of dry deposition, which did not start until the end of the twentieth century, is still limited, constraining our understanding of the dynamic processes of gas exchange at the land surface.

In contrast to wet deposition, dry deposition on a regional scale cannot be simply interpolated from point observations, because it is strongly affected by factors such as land surface properties (e.g., surface roughness) and climate (Erisman et al., 2005; Xu et al., 2015). Instead, the regional dry deposition has been estimated as the product of atmospheric N concentration and deposition velocities at the land surface (Flechard et al., 2011; Xu et al., 2015; Yu et al., 2019), where the N concentration can be interpolated or modeled from point observations, and the deposition velocities can be estimated based on modeled meteorological field and land cover map (Holland, Braswell, et al., 2005). However, different models and parameters used in estimating the two variables result in differences in the estimated dry deposition by a factor of 2–3 (Flechard et al., 2011). To improve the quantification of regional dry N deposition, the tropospheric column concentrations of NO_2 and NH_3 observed from satellites (e.g., Ozone Monitoring Instrument) have recently been used, in combination with ground observations, to infer NO_2 and NH_3 concentration at the land surface (Jia et al., 2016; Yu et al., 2019). While this advancement is promising to improve the spatial and temporal coverage of dry N deposition monitoring, more direct measurements of gaseous and particulate N fluxes over different land surfaces and in different climates are still needed to improve the understanding of the dynamic dry N deposition processes.

4.5. Nitrogen Fixation

N fixation is an important N budget term, estimated to account for about 17% of N inputs in the *Soil-Plant* system and about 32% in the *Agro-Food* system globally (Figure 4). Like denitrification, biological N fixation (BNF) is one of the most uncertain terms in N budgets, owing to the difficulty in measuring it. Several methods have been developed to attempt to measure BNF in a variety of croplands and natural ecosystems (Unkovich et al., 2008). Most plot-level estimates of N fixation by legume crops are based on a method that compares the isotopic signature of shoot tissue of the legume crop with a reference plant (often a nonleguminous “weed”) growing in the same soil (Högberg, 1997). The ^{15}N content of the reference plant reflects the isotopic signature of the available soil N pool, whereas N fixed from the atmosphere is assumed to have a delta ^{15}N signature of zero per mil or, in some cases, with a small fractionation correction factor of about −1 per mil (Alves et al., 2006). The percentage of N derived from the atmosphere (%Ndfa) is calculated from measurements of ^{15}N in the crop shoot tissue and a mixture model using the atmospheric inputs and the soil inputs as ^{15}N end-members. Estimates can be variable, but they frequently fall in the range of 60–80% of the N in shoots derived from the atmosphere for most leguminous crops (Anglade et al., 2015; Figueira et al., 2016; Peoples et al., 2009). The rate of N fixation is then calculated by multiplying this percentage by the total N stock of the shoots. A multiplier is then used to account for belowground N mass, which often adds another 30–70% to the BNF estimate and is another source of great uncertainty (Anglade et al., 2015).

The %Ndfa method and other methods have also been used to estimate associative N fixation by organisms living in the rhizosphere of some Gramineae crop plants, such as sugar cane, tropical pasture grasses, and rice (Unkovich et al., 2008). The estimated associative N fixation rates are generally lower than leguminous symbiotic BNF and remain somewhat controversial (James, 2000). A mass balance approach has been used to estimate nonsymbiotic BNF in maize, rice, and wheat production systems, where BNF was calculated from the difference of all of the other inputs and outputs (Ladha et al., 2016). The error term of this BNF estimate includes the errors of all of the other estimated inputs and output of the budget, including denitrification, which is also highly uncertain.

Labeling by incubating nodules in an atmosphere with $^{15}N_2$ is a more direct but costly method that is usually performed in the laboratory and then extrapolated to field rates based on estimates of nodule biomass and with assumptions relating laboratory and field conditions. Similarly, the activity of the nitrogenase enzyme can be estimated in laboratory or field incubations using acetylene reduction, adjusted by laboratory-calibrated conversion factors to rates of N_2 reduction, and then extrapolated to the plot scale (Cleveland et al., 1999).

Symbiotic BNF has been estimated in nonagricultural ecosystems by measuring N fixation rates with the acetylene reduction method and extrapolating those rates using estimates of plot-level nodule biomass or metrics on the abundance of leguminous tree species known to nodulate (Batterman et al., 2013; Sullivan et al., 2014). The %Ndfa method has also been applied to nonagricultural soils (Davidson et al., 2018). Fixation by free-living microorganisms in soil is usually a small but sometimes important input to nonagricultural ecosystems. It has usually been measured by the acetylene reduction method and scaled to ecosystem or even biome levels based on estimates of net primary productivity or evapotranspiration (Cleveland et al., 1999).

Regional and global bottom-up estimates of N fixation are usually based on assumed average rates of BNF by each leguminous crop and by scaling BNF in nonagricultural ecosystems using relationships with net primary productivity or evapotranspiration (e.g., Howarth et al., 2012). Mechanistic models of N fixation that include other environmental controls, such as nutrient limitations of N and P and temperature responses, have been applied at biome and global scales (Houlton et al., 2008). Mass balance and isotopic discrimination have been used to estimate BNF on a global scale (Vitousek et al., 2013). Despite these multiple approaches developed over decades, N fixation estimates remain poorly constrained in most N budgets.

4.6. Leaching and Runoff

Runoff and leaching are the major pathways that nutrients escape the agricultural land and contaminate water bodies. N and P in runoff and leaching can be measured directly in the field. To study nutrients in soil solution at different depths and its transport, there are sampling tools such as suction cups, lysimeters, ion-exchange resins, and groundwater wells (Lehmann & Schroth, 2003; Snyder, 1996). The nutrient loss in soil could also be measured using tracers, such as ^{32}P, ^{33}P, and ^{15}N (Lehmann & Schroth, 2003). To estimate N and P loss from cropland to water bodies for a larger scale (e.g., watershed scale), researchers can measure water flow (discharge) and nutrient concentration to determine the nutrient load, which includes discharge measurement and water sampling processes (Deelstra & Øygarden, 1998). When there are no discharge measurements, the annual rate can be estimated using climatological data (Deelstra et al., 1998). The choice of methods depends on factors such as project scale, project purpose, soil conditions, and cost (Lehmann & Schroth, 2003; Snyder, 1996).

Another way to estimate leaching and runoff is through modeling, especially for large-scale studies (e.g., global scale, Bouwman et al., 2017). The model can also be used to study how effective different best management practices (BMPs) are on reducing the nutrient loss (Borah & Bera, 2003). Different methodologies and different temporal and spatial scales have been adopted by different models, making it difficult to compare results across models and to evaluate uncertainties. For example, Mekonnen and Hoekstra (2018) estimate P load as the product of the P surplus of the *Soil-Plant* system and a spatially explicit erosion-runoff-leaching fraction; while the Integrated Model to Assess the Global Environment-Global Nutrient Model (Beusen et al., 2015; Bouwman et al., 2017) not only considers the P inputs as fertilizer and manure for the current season but also considers the P reserves in soil accumulated in previous years or decades. Comparing these and other modeling studies, the loads of global P in runoff and leaching to freshwater range from 0.16 to 5 Tg P $year^{-1}$ (Bouwman et al., 2009; Bouwman, Goldewijk, et al., 2013; Mekonnen & Hoekstra, 2018; Penuelas et al., 2013; Seitzinger et al., 2005).

Where direct measurements or models are not available, nutrient loss in leaching and runoff has been estimated with simple assumptions. For example, P in leaching is sometimes assumed as a fixed fraction of P budget terms, such as 12.5% of P inputs (Bouwman, Goldewijk, et al., 2013; Lun et al., 2018) or 20% of P surplus (Springmann et al., 2018). These simple assumptions introduce large uncertainties into results. For example, global P leaching and runoff from cropland estimated by Lun et al. (2018) is 20% lower than that by Bouwman, Goldewijk, et al. (2013), whereas the estimates for pasture by Lun et al. (2018) is 60% higher than Bouwman et al. (2009).

4.7. Gaseous Emission

While virtually no P exists in gaseous phase, N emission in the gaseous phase is an important part of the N budget for most systems. One of the major gaseous phase losses of N from terrestrial and aquatic ecosystems is in the form of N_2 gas as the end product of denitrification. Although N_2 emission does not have direct negative impacts on the environment, it is critical to quantify it for balancing the budget, and it represents N that might have been retained and recycled within the system for productive outputs. However, direct measurement of N_2 emission to the atmosphere from soils and water bodies has been challenging due to the high background concentrations in the atmosphere and high spatial and temporal variation in the emission (Davidson & Seitzinger, 2006). Attempts have been made to estimate N_2O:N_2 emission ratios in the laboratory and then apply an average ratio to field-based N_2O emission measurements to estimate N_2 emissions (Schlesinger, 2009), but the reported ratios are highly variable and the uncertainties using this approach are large due to several soil environmental factors (Houlton et al., 2013).

Ammonia (NH_3), nitric oxide (NO), and nitrous oxide (N_2O) are three other major gaseous emissions that negatively affect the environment (Davidson et al., 2011), including climate change and stratospheric ozone depletion (N_2O), downwind soil acidification (NO), formation of fine particulate matter that poses respiratory health risks (NH_3 and NO), and a precursor (NO) to formation of tropospheric ozone and nitrogen dioxide, which are also human respiratory health risks. With advances in gas analyzers and micrometeorological methods (e.g., soil chambers and micrometeorological towers), the emissions of all three gases have been measured on plot scales. Based on the plot-scale measurement, emission factors (e.g., IPCC, 2006), inventories (e.g., Davidson & Kanter, 2014; Houlton et al., 2013), and biogeochemical models (e.g., Del Grosso et al., 2005; Tian et al., 2019) have been developed to estimate emissions from agricultural and nonagricultural soils and from other sectors (e.g., industry and biomass burning) at regional to global scales. However, given large heterogeneity of emissions among locations, time periods, and management practices (Eagle et al., 2017), this upscaling, bottom-up approach inherently includes considerable uncertainties.

To evaluate and constrain those uncertainties, observations of NH_3 and N_2O emissions on regional and global scales have been developed. The global N_2O emission from anthropogenic activities has been derived from inverse modeling of temporal and spatial variation in atmospheric N_2O concentrations (top-down approach, Thompson et al., 2019; Prather et al., 2015). At broad global scales, the bottom-up and top-down approaches are in reasonable agreement, suggesting that agricultural sources account for two thirds or about 4.1 Tg N-N_2O year^{-1} of total anthropogenic emissions (Davidson & Kanter, 2014). However, the agreement between the two approaches has not always been reached on regional scales. For example, Griffis et al. (2013) and Zhang et al. (2014) found that the N_2O emission estimate from top-down atmospheric observation was about 2 times that of the estimate from bottom-up Intergovernmental Panel on Climate Change emission factors for United States corn belt region, suggesting underestimated emission factors for the region or unaccounted sources in this agriculture-dominated landscape. Top-down inverse modeling approaches are now being applied to continental and subcontinental scales (Thompson et al., 2019).

In addition to atmospheric measurement of gas concentration, advances in satellite observations provide another opportunity to validate and constrain NH_3 and NO_x emission on regional to global scale (Van Damme et al., 2018; Zhang et al., 2018). For example, a gridded NH_3 emission map was derived from the Infrared Atmospheric Sounding Interferometer satellite observation, and early results suggest that the bottom-up inventory (Emission Database for Global Atmospheric Research) may underestimate NH_3 emission by over 1 order of magnitude for many agricultural emission hot spots. Miyazaki et al. (2017) estimated global surface NO_x emissions from the assimilation of multiple satellite data sets, including tropospheric NO_2 columns, and profiles of O_3 and HNO_3.

4.8. Nutrient Storage or Depletion in Soil

Soil is one of the major nutrient stocks of a system that can increase or become depleted. Part of the P applied to crops has been shown to be retained in agricultural soils during multiple years of P application (Le Noë et al., 2018; Rowe et al., 2016; Sattari et al., 2014). This increased soil-P stock can become a source of P for future years, which is called the P legacy. This is the case of some European countries where decades of P overfertilization built up a large soil-P stock. Today, yields are often maintained even after a total elimination of P fertilization (Bouwman et al., 2017). Some soils, particularly those in tropical areas with highly weathered soils, are P-fixing because phosphate binds to iron and aluminum oxides (Roy et al., 2016). In the areas with high proportions of P-fixing soils, P recovery by crops can be low and P surpluses can be high, with most of the P surplus retained in the soil. The release of P fixed on iron and aluminum oxides may be slower than the release of P accumulating in less highly weathered soils in temperate regions. Therefore, the consideration of the P legacy effect is different when compared with soils that are “non-P-fixing.” P legacies can also play a role after agricultural activities cease, influencing P budgets in the postagricultural ecosystems (MacDonald et al., 2012).

The legacy effect is probably less common for N than for P, but N can also accumulate in soils. When soil N is at or near steady state, mineralization and immobilization of soil N are roughly in equilibrium. However, where soil C sequestration is occurring (e.g., conservation tillage), part of the N surplus will be retained with the C sequestration, promoting the stabilization of soil N through microbial turnover (Lin et al., 2016). van Groenigen et al. (2017) suggested that significant C sequestration at global scales (e.g., if the 4‰ initiative is achieved; Francaviglia et al., 2019) would consume a large fraction of N surplus. However, the N can also be lost if soil organic matter is later lost through erosion or net decomposition due to changing management practices. Van Meter et al. (2016) have introduced another N legacy concept, suggesting that N accumulated in agricultural soils during many years' fertilization can continue to affect stream and coastal water quality for several years to follow. This could be a contributing reason why the water quality in the Gulf of Mexico has not improved in the short term despite considerable mitigation efforts (Van Meter et al., 2018). A negative legacy effect results when N is mined from soils where inputs are insufficient (David et al., 2001) or where organic soils are drained (Van der Pol, 1992); soil fertility and productivity may decline as the result.

Changes of soil nutrients could be quantified by measuring nutrient contents in surface soil layers or soil profiles (Chen et al., 2017; Van Meter et al., 2016; Yan et al., 2014). However, the direct measurement approach faces major challenges to accurately quantify changes in soil nutrient stocks. Both nutrient concentrations (e.g., mg N g^{-1} dry soil) and bulk density (g dry soil cm^{-3} soil) must be accurately measured (Davidson & Ackerman, 1993). Unfortunately, bulk density measurements are commonly missing. Spatial heterogeneity of soil nutrient content is significant in both horizontal and vertical directions and can be affected by soil management practices such as tillage and fertilization (Christianson et al., 2012; Van Meter et al., 2016). Furthermore, short-term changes in nutrient stocks are usually small compared to the total stock (Yan et al., 2014). Therefore, the sampling error may be much larger than the actual change of nutrient stock in the soil. To quantify soil nutrient changes on plot to regional scales, more long-term site experiments and large-area soil sampling are needed, but both require significant human resources and financial support.

In addition to direct measurement of soil nutrient contents, isotope methods and modeling methods have been developed to estimate changes in soil nutrients. For example, Figueira et al. (2016) determined changes in inputs, outputs, and soil N stock using natural ^{15}N enrichment in the soils sampled from a chronosequence of soybean fields in Brazil. A simplified two-pool (labile and stable pools) P model, the Dynamic Phosphorus Pool Simulator (DPPS), has been developed to estimate P legacies from regional to global scales (Lun et al., 2018; Mogollón et al., 2018; Zhang et al., 2017). Other researchers used soil P dynamics models with more than two P pools (Goll et al., 2012; Ringeval et al., 2017; Wang et al., 2009; Yang et al., 2013) to study the distribution of soil P. Due to lack of data and understanding, it is difficult to compare and validate soil P pool estimates from different models.

5. Challenges in Informing Nutrient Management

The quantification of nutrient budgets has provided invaluable information for nutrient management. For example, the NUE and nutrient surplus for a *Soil-Plant* system on a farm scale can help farmers understand changes in soil nutrient stocks and estimate the fertilization needs for the next season. The quantification of nutrient footprints for the *Agro-Food* system informs individual consumers or institutions (e.g., a university) about the impacts of their day-to-day choices (e.g., dietary and energy choices) on nutrient pollution (Leach et al., 2016). However, using nutrient budgets to inform sustainable nutrient management faces multiple major challenges.

One of the major challenges is to define and assess "sustainable" nutrient management with nutrient budgets

Nutrient management affects multiple sustainability targets, involving complex trade-offs among stakeholders and across spatial scales. Nitrogen-efficient technologies on a farm scale do not necessarily lead to the reduction in nitrogen fertilizer use or nitrogen pollution for the region, given the complex feedbacks in the market and other socioeconomic factors (Zhang, Mauzerall, et al., 2015). Importing food from regions with higher NUE relieves nitrogen pollution in the importing countries and the world but adds the burden of more pollution to the regions exporting food (Huang et al., Oita et al., 2016; Zhang, 2017). Therefore, whether a nutrient management practice or food policy should be deemed as sustainable needs to be assessed on multiple spatial scales and is affected by the regional priorities in pursuing sustainability.

The quantification of nutrient budgets of various systems and spatial scales provides great opportunities for assessing impacts of nutrient management on multiple scales. Quantifying and connecting nutrient budgets across scales enables systematic evaluation and monitoring of nutrient management effectiveness. The importance of cross-scale assessments was clearly shown by Bai et al. (2014, 2018) for Chinese livestock systems. Increases in NUE were reported when estimated at the farm scale (herd scale) but when including the whole agro-food system NUE decreases were observed. Multi-system and multi-spatial scale approaches involving structural changes for alternative sustainable pathways at the system level are needed to improve sustainable use of manure by promoting reintegration of crop and livestock production systems (Garnier et al., 2016, Zhang, Liu, et al., 2019), to improve crop NUE through an optimal spatial allocation of fertilizers (Mueller et al., 2014, 2017; Zhang et al., 2018), to analyze the effect of dietary changes (Westhoek et al., 2014), or to combine supply- and demand-side actions (Bodirsky et al., 2014; Desmit et al., 2018; Lassaletta et al., 2016).

In addition to the cross-scale assessment, it is important to evaluate the impacts of nutrient management on other sustainability targets, such as food security and land use change. Integrating the efficiencies of nitrogen use (i.e., NUE) and land use (i.e., yield) on a national scale, a Sustainable Nitrogen Management Index has been developed and applied to measure the sustainability of agricultural production on the national scale (Sachs et al., 2016; Schmidt-Traub et al., 2017; Zhang & Davidson, 2019). However, this index is still limited in the scope of sustainability targets included in the assessment. To address this limitation, many ongoing efforts are devoted to develop indicators for assessing agriculture sustainability from environmental and socioeconomic perspectives and on various spatial scales (FAO, 2018b; Zhang, Liu, et al., 2019).

The second challenge is to connect actions by stakeholders with their impacts on the environment

Actions by stakeholders, such as consumers' dietary choices and farmers' choice of fertilizer management practices, are usually linked to the nutrient inputs and/or the efficiencies of the system directly; while the environmental impacts are best assessed by the damage to ecosystem and human health caused by nutrient loss in the form of pollutants. The environmental impacts are not only affected by the amount of nutrient loss quantified in a nutrient budget but also the timing and location of the loss, as well as many ecological factors, such as climate and soil conditions. Therefore, many environmental models have attempted to connect the stakeholder activities with the environmental impacts by simulating ecological processes involved in the emission, transformation, and transportation of the nutrient pollutants (Beusen et al., 2016; Desmit et al., 2018, Malagó et al., 2019, Schroeck et al., 2019). However, the wide adoption of environmental modeling approaches by stakeholders is limited by the regions or conditions where the models can be calibrated with data and by the transparency of the modeling inputs and processes (McLellan et al., 2018).

In contrast to the attempt in modeling the complex environmental impacts from nutrient budgets, several relatively simple indicators have been developed based on nutrient budgets. For example, N surplus (also called N balance) of the *Soil-Plant* system is directly affected by farmers' management practices and is proven to be significantly related to yield-scaled N losses when aggregated over multiple sites and years (McLellan et al., 2018; Sela et al., 2019). However, it should be recognized that the N surplus is not always the best predictor of the actual environmental impacts of an individual farm for a given year, which is also largely affected by factors such as year-to-year weather variations and the legacy effects of N in the soil. Similarly, N and P footprints are designed to connect consumers' dietary choices with broad-scale estimates of N and P pollution; however, most footprints are calculated with parameters generalized for a country and consequently limited in reflecting the different environmental impacts caused by different producers' choices in nutrient management practices. Where identifying the specific, near-term local impacts of the farmer or consumer decisions is desired, then more measurements and models tailored to those specific circumstances are needed.

The third major challenge is to effectively communicate nutrient budgets to engage actions

Simple and transparent messaging is critical for effective communication with stakeholders, especially for consumers. However, nutrient budgets, especially for N and P, are complicated with varying uncertainties, and the complex trade-offs among various sustainability goals and across scales further complicate calls for action. Consequently, a delicate balance needs to be reached to enable effective public communication with scientific rigor, and sometimes compromises need to be made considering the regional or global priorities.

Effective engagement of stakeholders for sustainable nutrient management should also go beyond the traditional “one-way” communication, namely from researchers to stakeholders. Involving stakeholders in the design and development of nutrient budgeting measurements and estimates and its communication products, such as indicators or decision-support models, is an important process for building trust and collaborative working relationships between researchers and stakeholders. It may also improve the accuracy of the nutrient budget terms and is a critical step toward engaging more stakeholders to incorporate science-based nutrient budget research in their decision making (Wall et al., 2017). However, few researchers and students have been trained to work effectively at the interface of science and communication (Safford et al., 2017), and the efforts in translating science to practices need to be better valued (Chapin, 2017).

Improving the accuracy of nutrient budgets is a worthy scientific pursuit, but communicating the meaning and significance of nutrient budgets for ecosystem and human health and their economic implications is paramount. Describing changes as accurately as possible in terms of teragrams (or any other units) of N or P per year is necessary for scientific integrity and to form the basis of a case for concern, but explaining the impact of changes in nutrient budgets goes beyond quantification of stocks and fluxes. Human alteration of the N budget has been reported by scientists for several decades (Crutzen, 1970; Davidson et al., 2011; Fowler et al., 2013; Galloway et al., 2003; Vitousek et al., 1997), but widespread appreciation within the policy community for the seriousness of the topic accelerated after the issue was described in terms of nitrogen excess exceeding a proposed planetary boundary of a “safe operating space for humanity” (Rockstrom et al., 2009). A crucial remaining challenge will require the convergence of biogeochemistry and agronomy with the social sciences in order to translate documented biophysical changes of nutrient budgets to quantitative or qualitative explanations of their impacts on economics, human health, and ecosystem functions that humans value.

6. Concluding Remarks

Managing nutrients is fundamental to sustainable development in this century, and it requires good information about nutrient budgets. The growing efforts in quantifying nutrient budgets for various systems and across different scales have enabled researchers to understand complex nutrient cycles and their interactions with human and Earth systems. Those efforts have also provided useful information for many stakeholders, including farmers, retailers, consumers, and policy makers, to assist their decisions that affect nutrient management. However, the expanding application still requires consistent and structural definitions of systems and their budget terms to enable comparisons and interlinkages among studies; large uncertainties in multiple nutrient budget terms need to be constrained; and the scientific findings need to be better communicated to engage positive changes.

By reviewing nutrient budget studies in five systems, namely *Soil-Plant* system, *Animal* system, *Animal-Plant-Soil* system, *Agro-Food* system, and *Landscape* system, this paper provides an initial attempt toward a consistent framework for nutrient budget analyses. The review of the quantification challenges reveals the varying uncertainties in major nutrient budget terms across spatial scales. The levels of challenge in constraining those uncertainties also vary largely among budget terms and scales: some already have well-established methods for measuring and validating, while others still rely on very crude parameters developed from few studies or observations and require the development of basic observation approaches. To address the quantification challenges, advances in monitoring both ecological and socioeconomic changes are needed. Recent developments in remote sensing, sensor technologies, and isotope methods provide opportunities to overcome difficulties in monitoring nutrient flows on a variety of spatial and temporal scales. Increasing interest in quantifying nutrient budgets by governments and farmers could potentially motivate more resources toward improving the collection of human activity data related to nutrient budgets (e.g., fertilization rate and food trade). To take advantage of those opportunities, interdisciplinary modeling and data synthesis approaches need to be developed.

The current limitations in nutrient budgets and the need for their improvement should be better communicated not only among researchers but also stakeholders engaged in nutrient management. It will take decades, even centuries, for all nutrient budget terms to be accurately accounted for, but this should not prevent scientists from communicating the importance of nutrient budgets and the insights they provide for sustainable management.

Acknowledgments
This work was supported by National Science Foundation CNS-1739823 and National Socio-Environmental Synthesis Center (SESYNC) under funding received from the National Science Foundation DBI-1639145. L. Lassaletta is supported by MINECO-Spain and European Commission ERDF Ramón y Cajal grant (RYC-2016-20269), Programa Propio from UPM, and acknowledges the Comunidad de Madrid (Spain) and structural funds 2014-2020 (ERDF and ESF), project AGRISOST-CM S2018/BAA-4330. We thank Dylan Taillie at the University of Maryland Center for Environmental Science Appalachian Laboratory for improving the visualization of Figure 3. Data are shown in Figures 4–7: https://doi.org/10.5061/dryad.q83bk3jf3.

References

Allison, F. E. (1955). The enigma of soil nitrogen balance sheets. In *Advances in Agronomy* (Vol. 7, pp. 213–250). Cambridge, MA: Elsevier. https://doi.org/10.1016/S0065-2113

Alves, B. J. R., Zotarelli, L., Fernandes, F. M., Heckler, J. C., Macedo, R. A. T. D., Boddey, R. M., et al. (2006). Biological nitrogen fixation and nitrogen fertilizer on the nitrogen balance of soybean, maize and cotton. *Pesquisa Agropecuária Brasileira*, *41*(3), 449–456. https://doi.org/10.1590/S0100-204X2006000300011

Anglade, J., Billen, G., & Garnier, J. (2015). Relationships for estimating N_2 fixation in legumes: Incidence for N balance of legume-based cropping systems in Europe. *Ecosphere*, *6*(3), 1–24. https://doi.org/10.1890/ES14-00353.1

AUSNUT. (2013). AUSNUT 2011-13 food nutrient database. Retrieved from http://www.foodstandards.gov.au/science/monitoringnutrients/ausnut/ausnutdatafiles/Pages/foodnutrient.aspx

Bai, Z., Ma, L., Qin, W., Chen, Q., Oenema, O., & Zhang, F. (2014). Changes in pig production in China and their effects on nitrogen and phosphorus use and losses. *Environmental Science & Technology*, *48*(21), 12,742–12,749. https://doi.org/10.1021/es502160v

Bai, Z., Ma, W., Ma, L., Velthof, G. L., Wei, Z., Havlik, P., et al. (2018). China's livestock transition: Driving forces, impacts, and consequences. *Science Advances*, *4*(7), eaar8534. https://doi.org/10.1126/sciadv.aar8534

Batterman, S. A., Hedin, L. O., Van Breugel, M., Ransijn, J., Craven, D. J., & Hall, J. S. (2013). Key role of symbiotic dinitrogen fixation in tropical forest secondary succession. *Nature*, *502*(7470), 224–227. https://doi.org/10.1038/nature12525

Beusen, A. H., Bouwman, A. F., Van Beek, L. P., Mogollón, J. M., & Middelburg, J. J. (2016). Global riverine N and P transport to ocean increased during the 20th century despite increased retention along the aquatic continuum. *Biogeosciences*, *13*(8), 2441–2451. https://doi.org/10.5194/bg-13-2441-2016

Beusen, A. H. W., Van Beek, L. P. H., Bouwman, A. F., Mogollón, J. M., & Middelburg, J. J. (2015). Coupling global models for hydrology and nutrient loading to simulate nitrogen and phosphorus retention in surface water—Description of IMAGE-GNM and analysis of performance. *Geoscientific Model Development Discussion*, *8*(9), 7477–7539. https://doi.org/10.5194/gmd-8-4045-2015

Billen, G., Lassaletta, L., & Garnier, J. (2014). A biogeochemical view of the global agro-food system: Nitrogen flows associated with protein production, consumption and trade. *Global Food Security*, *3*(3), 209–219. https://doi.org/10.1016/j.gfs.2014.08.003

Bodirsky, B. L., Popp, A., Lotze-Campen, H., Dietrich, J. P., Rolinski, S., Weindl, I., et al. (2014). Reactive nitrogen requirements to feed the world in 2050 and potential to mitigate nitrogen pollution. *Nature Communications*, *5*(1), 3858. https://doi.org/10.1038/ncomms4858

Bodirsky, B. L., Popp, A., Weindl, I., Dietrich, J. P., Rolinski, S., Scheiffele, L., et al. (2012). N_2O emissions from the global agricultural nitrogen cycle–current state and future scenarios. *Biogeosciences*, *9*(10), 4169–4197. https://doi.org/10.5194/bg-9-4169-2012

Borah, D. K., & Bera, M. (2003). Watershed-scale hydrologic and nonpoint-source pollution models: Review of mathematical bases. *Transactions of ASAE*, *46*(6), 1553. Retrieved from https://pdfs.semanticscholar.org/ff1d/58c825d301a82b22db8db6352308b0877ccf.pdf

Bormann, F. H., & Likens, G. E. (1979). *Pattern and process in a forested ecosystem: Disturbance, development, and the steady state based on the Hubbard Brook ecosystem study*. Verlag: Springer. https://doi.org/10.1017/S0376892900028320

Boussingault, J. (1841). De la descussion de la valeur relative des assolements, par les resultats de l'analyse elementaire. *Annales de Chimie Physique*, *1*, 208–246. https://doi.org/10.2307/27849584

Bouwman, A., Beusen, A., Griffioen, J., Van Groenigen, J., Hefting, M., Oenema, O., et al. (2013). Global trends and uncertainties in terrestrial denitrification and N_2O emissions. *Philosophical Transactions of the Royal Society, B: Biological Sciences*, *368*(1621), 20130112. https://doi.org/10.1098/rstb.2013.0112

Bouwman, A. F. (1997). *Long-term scenarios of livestock-crop-land use interactions in developing countries*. Rome, Italy: Food & Agriculture Org.

Bouwman, A. F., Beusen, A. H., Lassaletta, L., van Apeldoorn, D. F., van Grinsven, H. J., Zhang, J., & van Ittersum, M. K. (2017). Lessons from temporal and spatial patterns in global use of N and P fertilizer on cropland. *Scientific Reports*, *7*(1), 40366. https://doi.org/10.1038/srep40366

Bouwman, A. F., Beusen, A. H. W., & Billen, G. (2009). Human alteration of the global nitrogen and phosphorus soil balances for the period 1970–2050. *Global Biogeochemical Cycles*, *23*, GB0A04. https://doi.org/10.1029/2009GB003576

Bouwman, L., Goldewijk, K. K., Van Der Hoek, K. W., Beusen, A. H., Van Vuuren, D. P., Willems, J., et al. (2013). Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900–2050 period. *Proceedings of the National Academy of Sciences*, *110*(52), 20,882–20,887. https://doi.org/10.1073/pnas.1012878108

Boyer, E. W., Goodale, C. L., Jaworski, N. A., & Howarth, R. W. (2002). Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern U.S.A. *Biogeochemistry*, *57*(1), 137–169. https://doi.org/10.1023/a:1015709302073

Cao, P., Lu, C., & Yu, Z. (2018). Historical nitrogen fertilizer use in agricultural ecosystems of the contiguous United States during 1850–2015: Application rate, timing, and fertilizer types. *Earth System Science Data*, *10*(2), 969. https://doi.org/10.5194/essd-10-969-2018

Chapin, F. S. III (2017). Now is the time for translational ecology. *Frontiers in Ecology and the Environment*, *15*(10), 539–539. https://doi.org/10.1002/fee.1737

Chen, D., Hu, M., Guo, Y., Wang, J., Huang, H., & Dahlgren, R. A. (2017). Long-term (1980–2010) changes in cropland phosphorus budgets, use efficiency and legacy pools across townships in the Yongan watershed, eastern China. *Agriculture, Ecosystems & Environment*, *236*, 166–176. https://doi.org/10.1016/j.agee.2016.12.003

Christianson, L., Castellano, M., & Helmers, M. (2012). Nitrogen and phosphorus balances in Iowa cropping systems: sustaining Iowa's soil resource. Legislative Report. 66191. Retrieved from https://www.legis.iowa.gov

Ciampitti, I. A., & Vyn, T. J. (2012). Physiological perspectives of changes over time in maize yield dependency on nitrogen uptake and associated nitrogen efficiencies: A review. *Field Crops Research*, *133*, 48–67. https://doi.org/10.1016/j.fcr.2012.03.008

Cleveland, C. C., Townsend, A. R., Schimel, D. S., Fisher, H., Howarth, R. W., Hedin, L. O., et al. (1999). Global patterns of terrestrial biological nitrogen (N_2) fixation in natural ecosystems. *Global Biogeochemical Cycles*, *13*(2), 623–645. https://doi.org/10.1029/1999GB900014

Cooper, J., Lombardi, R., Boardman, D., & Carliell-Marquet, C. (2011). The future distribution and production of global phosphate rock reserves. *Resources, Conservation and Recycling*, *57*, 78–86. https://doi.org/10.1016/j.resconrec.2011.09.009

Cordell, D., Drangert, J.-O., & White, S. (2009). The story of phosphorus: Global food security and food for thought. *Global Environmental Change*, *19*(2), 292–305. https://doi.org/10.1016/j.gloenvcha.2008.10.009

Cordell, D., & White, S. (2014). Life's bottleneck: sustaining the world's phosphorus for a food secure future. *Annual Review of Environment and Resources*, *39*, 161–188. https://doi.org/10.1146/annurev-environ-010213-113300

Cordell, D., White, S., & Lindström, T. (2011). Peak phosphorus: The crunch time for humanity. *The Sustainability Review*, *2*(2), 3. Retrieved from https://www.thesustainabilityreview.org/articles/

Crutzen, P. J. (1970). The influence of nitrogen oxides on the atmospheric ozone content. *Quarterly Journal of the Royal Meteorological Society*, *96*(408), 320–325. https://doi.org/10.1002/qj.49709640815

Crutzen, P. J., Mosier, A. R., Smith, K. A., & Winiwarter, W. (2008). N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. *Atmospheric Chemistry and Physics*, *8*(2), 389–395. https://doi.org/10.5194/acp-8-389-2008

David, M. B., McIsaac, G. F., Royer, T. V., Darmody, R. G., & Gentry, L. E. (2001). Estimated historical and current nitrogen balances for Illinois. *The Scientific World Journal*, *1*(Suppl 2), 597–604. https://doi.org/10.1100/tsw.2001.283

Davidson, E. A. (2009). The contribution of manure and fertilizer nitrogen to atmospheric nitrous oxide since 1860. *Nature Geoscience*, *2*(9), 659–662. https://doi.org/10.1038/ngeo608

Davidson, E. A., & Ackerman, I. L. (1993). Changes in soil carbon inventories following cultivation of previously untilled soils. *Biogeochemistry*, *20*(3), 161–193. https://doi.org/10.2307/1469217

Davidson, E. A., David, M. B., Galloway, J. N., Goodale, C. L., Haeuber, R., Harrison, J. A., et al. (2011). Excess nitrogen in the U.S. environment: Trends, risks, and solutions. *Issues in Ecology*, (15).

Davidson, E. A., & Kanter, D. (2014). Inventories and scenarios of nitrous oxide emissions. *Environmental Research Letters*, *9*(10), 105012. https://doi.org/10.1088/1748-9326/9/10/105012

Davidson, E. A., Markewitz, D., de O. Figueiredo, R., & de Camargo, P. B. (2018). Nitrogen fixation inputs in pasture and early successional forest in the Brazilian amazon region: Evidence from a claybox mesocosm study. *Journal of Geophysical Research: Biogeosciences*, *123*, 712–721. https://doi.org/10.1002/2017JG004103

Davidson, E. A., Nifong, R. L., Ferguson, R. B., Palm, C., Osmond, D. L., & Baron, J. S. (2016). Nutrients in the nexus. *Journal of Environmental Studies and Sciences*, *6*(1), 25–38. https://doi.org/10.1007/s13412-016-0364-y

Davidson, E. A., & Seitzinger, S. (2006). The enigma of progress in denitrification research. *Ecological Applications*, *16*(6), 2057–2063. https://doi.org/10.1890/1051-0761(2006)016[2057:TEOPID]2.0.CO;2

Deelstra, J., & Øygarden, L. (1998). Measurement of runoff. In L. Øygarden & P. Botterweg (Eds.), *Measuring runoff and nutrient loss from agricultural land in Nordic countries* (pp. 13–26). Nordic Council of Ministers: Copenhagen.

Deelstra, J., Vagstad, N., & Øygarden, L. (1998). Sampling technique and strategy. In L. Øygarden & P. Botterweg (Eds.), *Measuring runoff and nutrient loss from agricultural land in Nordic countries* (pp. 27–35). Nordic Council of Ministers: Copenhagen.

Del Grosso, S. J., Mosier, A. R., Parton, W. J., & Ojima, D. S. (2005). DAYCENT model analysis of past and contemporary soil N_2O and net greenhouse gas flux for major crops in the USA. *Soil and Tillage Research*, *83*(1), 9–24. https://doi.org/10.1016/j.still.2005.02.007

Deming, W. E. (2018). *The new economics for industry, government, education*. Cambridge, MA: MIT Press.

Desmit, X., Thieu, V., Billen, G., Campuzano, F., Dulière, V., Garnier, J., et al. (2018). Reducing marine eutrophication may require a paradigmatic change. *Science of the Total Environment*, *635*, 1444–1466. https://doi.org/10.1016/j.scitotenv.2018.04.181

de Vries, W., Leip, A., Reinds, G. J., Kros, J., Lesschen, J. P., & Bouwman, A. F. (2011). Comparison of land nitrogen budgets for European agriculture by various modeling approaches. *Environmental Pollution*, *159*, 3254–3268.

Dong, H., Mangino, J., McAllister, T. A., Hatfield, J. L., Johnson, D. E., Lassey, K. R., et al. (2006). Chapter 10. Emissions from livestock and manure management. In *2006 IPCC guidelines for national greenhouse gas inventories* (Vol. *4*, pp. 10.1–10.87). IGES, Japan: Institute for Global Environmental Strategies.

Dourado, L. R. B., Pascoal, L. A. F., Sakomura, N. K., Costa, F. G. P., & Biagiotti, D. (2011). Soybeans (Glycine max) and soybean products in poultry and swine nutrition. In *Recent trends for enhancing the diversity and quality of soybean products* (pp. 175–190). London, UK: IntechOpen. https://doi.org/10.5772/18071

Eagle, A. J., Olander, L. P., Locklier, K. L., Heffernan, J. B., & Bernhardt, E. S. (2017). Fertilizer management and environmental factors drive N_2O and NO_3 losses in corn: A meta-analysis. *Soil Science Society of America Journal*, *81*(5), 1191–1202. https://doi.org/10.2136/sssaj2016.09.0281

Edixhoven, J. D., Gupta, J., & Savenije, H. H. G. (2014). Recent revisions of phosphate rock reserves and resources: A critique. *Earth System Dynamics*, *5*(2), 491–507. https://doi.org/10.5194/esd-5-491-2014

Ehhalt, D., Prather, M., Dentener, F., Derwent, R., Dlugokencky, E. J., Holland, E., et al. (2001). Atmospheric chemistry and greenhouse gases. Retrieved from https://www.osti.gov/biblio/901482

Erisman, J., Leach, A., Bleeker, A., Atwell, B., Cattaneo, L., & Galloway, J. (2018). An integrated approach to a nitrogen use efficiency (NUE) indicator for the food production–consumption chain. *Sustainability*, *10*(4), 925. https://doi.org/10.3390/su10040925

Erisman, J. W., Vermeulen, A., Hensen, A., Flechard, C., Dämmgen, U., Fowler, D., et al. (2005). Monitoring and modelling of biosphere/atmosphere exchange of gases and aerosols in Europe. *Environmental Pollution*, *133*(3), 403–413. https://doi.org/10.1016/j.envpol.2004.07.004

EUNEP (2015). *Nitrogen use efficiency (NUE)—An indicator for the utilization of nitrogen in agriculture and food systems*. Wageningen University, Alterra, Wageningen, Netherlands: EU Nitrogen Expert Panel.

FAO. (2006). Plant nutrition for food security: A guide for integrated nutrient management. Retrieved from http://www.fao.org/3/a0443e/a0443e00.htm

FAO. (2018a). Nitrogen inputs to agricultural soils from livestock manure. New Statistics Retrieved from http://www.fao.org/documents/card/en/c/I8153EN/

FAO. (2018b). SDG 2.4.1 Proportion of agricultural area under productive and sustainable agriculture.

FAOSTAT. (2019). Online databases. Retrieved from: http://www.fao.org/faostat/en/Feedipedia. (2012-2017). Animal feed resources information system. Retrieved from https://feedipedia.org/

Feedipedia. (2012–2017). Animal feed resources information system. Retrieved from https://feedipedia.org/

Fenn, M. E., Jovan, S., Yuan, F., Geiser, L., Meixner, T., & Gimeno, B. S. (2008). Empirical and simulated critical loads for nitrogen deposition in California mixed conifer forests. *Environmental Pollution*, *155*(3), 492–511. https://doi.org/10.1016/j.envpol.2008.03.019

Figueira, A. M. E. S., Davidson, E. A., Nagy, R. C., Riskin, S. H., & Martinelli, L. A. (2016). Isotopically constrained soil carbon and nitrogen budgets in a soybean field chronosequence in the Brazilian Amazon region. *Journal of Geophysical Research: Biogeosciences*, *121*, 2520–2529. https://doi.org/10.1002/2016JG003470

Fixen, P. E., Williams, R., & Rund, Q. B. (2012). *NUGIS: A nutrient use geographic information system for the US*. Brookings, SD. Retrieved from: International Plant Nutrition Institute. http://nugis.ipni.net/Publication/

Flechard, C., Nemitz, E., Smith, R., Fowler, D., Vermeulen, A., Bleeker, A., et al. (2011). Dry deposition of reactive nitrogen to European ecosystems: A comparison of inferential models across the NitroEurope network. *Atmospheric Chemistry and Physics*, *11*(6), 2703–2728. https://doi.org/doi:10.5194/acp-11-2703-2011

Fowler, D., Coyle, M., Skiba, U., Sutton, M. A., Cape, J. N., Reis, S., et al. (2013). The global nitrogen cycle in the twenty-first century. *Philosophical Transactions of the Royal Society, B: Biological Sciences*, *368*(1621), 20130164. https://doi.org/10.1098/rstb.2013.0164

Francaviglia, R., Di Bene, C., Farina, R., Salvati, L., & Vicente-Vicente, J. L. (2019). Assessing "4 per 1000" soil organic carbon storage rates under Mediterranean climate: A comprehensive data analysis. *Mitigation and Adaptation Strategies for Global Change*, *24*(5), 795–818. https://doi.org/10.1007/s11027-018-9832-x

Galloway, J. N., Aber, J. D., Erisman, J. W., Seitzinger, S. P., Howarth, R. W., Cowling, E. B., & Cosby, B. J. (2003). The nitrogen cascade. *Bioscience*, *53*(4), 341–356. <Go to ISI>://000182254000011

Galloway, J. N., Leach, A. M., Bleeker, A., & Erisman, J. W. (2013). A chronology of human understanding of the nitrogen cycle. *Philosophical Transactions of the Royal Society, B: Biological Sciences*, *368*(1621), 20130120. https://doi.org/10.1098/rstb.2013.0120

Garnier, J., Anglade, J., Benoit, M., Billen, G., Puech, T., Ramarson, A., et al. (2016). Reconnecting crop and cattle farming to reduce nitrogen losses to river water of an intensive agricultural catchment (Seine basin, France): Past, present and future. *Environmental Science & Policy*, *63*, 76–90. https://doi.org/10.1016/j.envsci.2016.04.019

Garnier, J., Beusen, A., Thieu, V., Billen, G., & Bouwman, L. (2010). N: P: Si nutrient export ratios and ecological consequences in coastal seas evaluated by the ICEP approach. *Global Biogeochemical Cycles*, *24*(4), GB0A05. https://doi.org/10.1029/2009GB003583

GFSAD. (2019). Global Croplands. Retrieved from https://croplands.org/app/map/statsMap

Glibert, P. M., Icarus Allen, J., Artioli, Y., Beusen, A., Bouwman, L., Harle, J., et al. (2014). Vulnerability of coastal ecosystems to changes in harmful algal bloom distribution in response to climate change: Projections based on model analysis. *Global Change Biology*, *20*(12), 3845–3858. https://doi.org/10.1111/gcb.12662

Godinot, O., Leterme, P., Vertès, F., Faverdin, P., & Carof, M. (2015). Relative nitrogen efficiency, a new indicator to assess crop livestock farming systems. *Agronomy for Sustainable Development*, *35*(2), 857–868. https://doi.org/10.1007/s13593-015-0281-6

Goll, D. S., Brovkin, V., Parida, B. R., Reick, C. H., Kattge, J., Reich, P. B., et al. (2012). Nutrient limitation reduces land carbon uptake in simulations with a model of combined carbon, nitrogen and phosphorus cycling. *Biogeosciences*, *9*(3), 3173–3232. https://doi.org/10.5194/bgd-9-3173-2012

Gourley, C., Dougherty, W., Aarons, S., & Hannah, M. (2010). Accounting for nutrients on Australian dairy farms. Final Report. Victorian Government Department of Primary Industries, Ellinbank, Vic. Retrieved from http://ecite.utas.edu.au/53560

Granat, L., Rodhe, H., & Hallberg, R. O. (1976). The global sulphur cycle. *Ecological Bulletins*, *22*, 89–134. Retrieved from http://www.jstor.org/stable/20112523

Griffis, T. J., Lee, X., Baker, J. M., Russelle, M. P., Zhang, X., Venterea, R., & Millet, D. B. (2013). Reconciling the differences between top-down and bottom-up estimates of nitrous oxide emissions for the U.S. Corn Belt. *Global Biogeochemical Cycles*, *27*(3), 746–754. https://doi.org/10.1002/gbc.20066

Groffman, P. M., Driscoll, C. T., Durán, J., Campbell, J. L., Christenson, L. M., Fahey, T. J., et al. (2018). Nitrogen oligotrophication in northern hardwood forests. *Biogeochemistry*, *141*(3), 523–539. https://doi.org/10.1007/s10533-018-0445-y

Guardia, G., Sanz-Cobena, A., Sanchez-Martín, L., Fuertes-Mendizábal, T., González-Murua, C., Álvarez, J. M., et al. (2018). Urea-based fertilization strategies to reduce yield-scaled N oxides and enhance bread-making quality in a rainfed Mediterranean wheat crop. *Agriculture, Ecosystems & Environment*, *265*, 421–431. https://doi.org/10.1016/j.agee.2018.06.033

Han, Y., Fan, Y., Yang, P., Wang, X., Wang, Y., Tian, J., et al. (2014). Net anthropogenic nitrogen inputs (NANI) index application in Mainland China. *Geoderma*, *213*, 87–94. https://doi.org/10.1016/j.geoderma.2013.07.019

Heffer, P. (2013). *Assessment of fertilizer use by crop at the global level 2010–2010/11*. Paris: International Fertilizer Industry Association. http://www.fertilizer.org/En/Statistics/Agriculture_Committee_Databases.aspx

Heffer, P., Gruère, A., & Roberts, T. (2017). *Assessment of fertilizer use by crop at the global level 2014–2014/15*. Paris, Retrieved from: International Fertilizer Industry Association. http://www.fertilizer.org//En/Statistics/Agriculture_Committee_Databases.aspx

Hember, R. A. (2018). Spatially and temporally continuous estimates of annual total nitrogen deposition over North America, 1860–2013. *Data in Brief*, *17*, 134–140. https://doi.org/10.1016/j.dib.2017.12.052

Hoekstra, A. Y., & Wiedmann, T. O. (2014). Humanity's unsustainable environmental footprint. *Science*, *344*(6188), 1114–1117. https://doi.org/10.1126/science.1248365

Högberg, P. (1997). Tansley Review No. 95 ^{15}N natural abundance in soil-plant systems. *New Phytologist*, *137*(2), 179–203. https://doi.org/10.1046/j.1469-8137.1997.00808.x

Holland, E. A., Braswell, B. H., Sulzman, J., & Lamarque, J.-F. (2005). Nitrogen deposition onto the United States and Western Europe: Synthesis of observations and models. *Ecological Applications*, *15*(1), 38–57. https://doi.org/10.1890/03-5162

Holland, E. A., Lee-Taylor, J., Nevison, C., & Sulzman, J. M. (2005). Global N cycle: Fluxes and N_2O mixing ratios originating from human activity. *ORNL DAAC*, Oak Ridge, Tennessee, USA. https://doi.org/10.3334/ORNLDAAC/797

Hong, B., Swaney, D. P., & Howarth, R. W. (2013). Estimating net anthropogenic nitrogen inputs to US watersheds: Comparison of methodologies. *Environmental Science & Technology*, *47*(10), 5199–5207. https://doi.org/10.1021/es303437c

Hong, B., Swaney, D. P., Mörth, C.-M., Smedberg, E., Hägg, H. E., Humborg, C., et al. (2012). Evaluating regional variation of net anthropogenic nitrogen and phosphorus inputs (NANI/NAPI), major drivers, nutrient retention pattern and management implications in the multinational areas of Baltic Sea basin. *Ecological Modelling*, *227*, 117–135. https://doi.org/10.1016/j.ecolmodel.2011.12.002

Hou, Y., Bai, Z., Lesschen, J. P., Staritsky, I. G., Sikirica, N., Ma, L., et al. (2016). Feed use and nitrogen excretion of livestock in EU-27. *Agriculture, Ecosystems & Environment*, *218*, 232–244. https://doi.org/10.1016/j.agee.2015.11.025

Houlton, B. Z., Almaraz, M., Aneja, V., Austin, A. T., Bai, E., Cassman, K. G., et al. (2019). A world of co-benefits: Solving the global nitrogen challenge. *Earth's Future.*, *7*(8), 865–872. https://doi.org/10.1029/2019EF001222

Houlton, B. Z., Boyer, E., Finzi, A., Galloway, J., Leach, A., Liptzin, D., et al. (2013). Intentional versus unintentional nitrogen use in the United States: Trends, efficiency and implications. *Biogeochemistry*, *114*(1-3), 11–23. https://doi.org/10.1007/s10533-012-9801-5

Houlton, B. Z., Wang, Y.-P., Vitousek, P. M., & Field, C. B. (2008). A unifying framework for dinitrogen fixation in the terrestrial biosphere. *Nature*, *454*(7202), 327–330. https://doi.org/10.1038/nature07028

Howarth, R., Swaney, D., Billen, G., Garnier, J., Hong, B., Humborg, C., et al. (2012). Nitrogen fluxes from the landscape are controlled by net anthropogenic nitrogen inputs and by climate. *Frontiers in Ecology and the Environment*, *10*(1), 37–43. https://doi.org/10.1890/100178

Howarth, R. W., Billen, G., Swaney, D., Townsend, A., Jaworski, N., Lajtha, K., et al. (1996). Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. In *Nitrogen cycling in the North Atlantic Ocean and its watersheds* (pp. 75–139). Dordrecht, Netherlands: Springer. https://doi.org/10.1007/BF02179825

Huang, G., Yao, G., Zhao, J., Lisk, M. D., Yu, C., & Zhang, X. (2019). The environmental and socioeconomic trade-offs of importing crops to meet domestic food demand in China. *Environmental Research Letters*.

IFA. (2019). IFA DATA. Retrieved from http://ifadata.fertilizer.org/ucSearch.aspx

IPCC. (2006). 2006 IPCC guidelines for national greenhouse gas inventories. Retrieved from IGES, Japan:

IPNI. (2014). IPNI Estimates of Nutrient Uptake and Removal Retrieved from Retrieved from http://www.ipni.net/article/IPNI-3296

James, E. K. (2000). Nitrogen fixation in endophytic and associative symbiosis. *Field Crops Research*, *65*(2-3), 197–209. https://doi.org/10.1016/s0378-4290(99)00087-8

Jia, Y., Yu, G., Gao, Y., He, N., Wang, Q., Jiao, C., & Zuo, Y. (2016). Global inorganic nitrogen dry deposition inferred from ground-and space-based measurements. *Scientific Reports*, *6*, 19810. https://doi.org/10.1038/srep19810

Jordan, T. E., & Weller, D. E. (1996). Human contributions to terrestrial nitrogen flux. *Bioscience*, *46*(9), 655–664. https://doi.org/10.2307/1312895

Khabarov, N., & Obersteiner, M. (2017). Global phosphorus fertilizer market and national policies: A case study revisiting the 2008 price peak. *Frontiers in Nutrition*, *4*, 22. https://doi.org/10.2307/1312895

Ladha, J. K., Tirol-Padre, A., Reddy, C. K., Cassman, K. G., Verma, S., Powlson, D. S., et al. (2016). Global nitrogen budgets in cereals: A 50-year assessment for maize, rice, and wheat production systems. *Scientific Reports*, *4*, 19355. https://doi.org/10.3389/fnut.2017.00022

Lassaletta, L., Billen, G., Garnier, J., Bouwman, L., Velazquez, E., Mueller, N. D., & Gerber, J. S. (2016). Nitrogen use in the global food system: Past trends and future trajectories of agronomic performance, pollution, trade, and dietary demand. *Environmental Research Letters*, *11*(9), 095007. https://doi.org/10.1088/1748-9326/11/9/095007

Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J., & Garnier, J. (2014). 50 year trends in nitrogen use efficiency of world cropping systems: The relationship between yield and nitrogen input to cropland. *Environmental Research Letters*, *9*(10), 105011. https://doi.org/10.1088/1748-9326/9/10/105011

Lassaletta, L., Billen, G., Grizzetti, B., Garnier, J., Leach, A. M., & Galloway, J. N. (2014). Food and feed trade as a driver in the global nitrogen cycle: 50-year trends. *Biogeochemistry*, *118*(1-3), 225–241. https://doi.org/10.1007/s10533-013-9923-4

Lassaletta, L., Estelles, F., Beusen, A. H. W., Lex Bouwman, L., Calvet, S., van Grinsven, H. J. M., et al. (2019). Future global pig production systems according to the shared socioeconomic pathways. *Science of the Total Environment*, *665*, 739–751. https://doi.org/10.1016/j.scitotenv.2019.02.079

Lassaletta, L., Romero, E., Billen, G., Garnier, J., García-Gómez, H., & Rovira, J. (2012). Spatialized N budgets in a large agricultural Mediterranean watershed: High loading and low transfer. *Biogeosciences*, *9*(1), 57–70. https://doi.org/10.5194/bg-9-57-2012

Lawes, J. B. (1882). *On the amount and composition of the rain and drainage-waters collected at Rothamsted*. London: William Clowes and Sons. Limited

Le Noë, J., Billen, G., & Garnier, J. (2017). How the structure of agro-food systems shapes nitrogen, phosphorus, and carbon fluxes: The generalized representation of agro-food system applied at the regional scale in France. *Science of the Total Environment*, *586*, 42–55. https://doi.org/10.1016/j.scitotenv.2017.02.040

Le Noë, J., Garnier, J., & Billen, G. (2018). Phosphorus management in cropping systems of the Paris Basin: From farm to regional scale. *Journal of Environmental Management*, *205*, 18–28. https://doi.org/10.1016/j.jenvman.2017.09.039

Leach, A. M., Emery, K. A., Gephart, J., Davis, K. F., Erisman, J. W., Leip, A., et al. (2016). Environmental impact food labels combining carbon, nitrogen, and water footprints. *Food Policy*, *61*, 213–223. https://doi.org/10.1016/j.foodpol.2016.03.006

Leach, A. M., Galloway, J. N., Bleeker, A., Erisman, J. W., Kohn, R., & Kitzes, J. (2012). A nitrogen footprint model to help consumers understand their role in nitrogen losses to the environment. *Environmental Development*, *1*(1), 40–66. https://doi.org/10.1016/j.envdev.2011.12.005

Lehmann, J., & Schroth, G. (2003). Nutrient Leaching. In G. Schroth & F. L. Sinclair (Eds.), *Trees, crops, and soil fertility: Concepts and research methods* (pp. 151–166). Wallingford, UK: CABI. https://doi.org/10.1079/9780851995939.0151

Leip, A., Britz, W., Weiss, F., & de Vries, W. (2011). Farm, land, and soil nitrogen budgets for agriculture in Europe calculated with CAPRI. *Environmental Pollution*, *159*(11), 3243–3253. https://doi.org/10.1016/j.envpol.2011.01.040

Leip, A., Leach, A., Musinguzi, P., Tumwesigye, T., Olupot, G., Stephen Tenywa, J., et al. (2014). Nitrogen-neutrality: a step towards sustainability. *Environmental Research Letters*, *9*(11), 115001. https://doi.org/10.1088/1748-9326/9/11/115001

Li, T., Zhang, X., Davidson, E. A., Dou, Z., Zhang, W., & Zhang, F. (2019). A hierarchical framework for unpacking the nitrogen challenge. *Earth and Space Science Open Archive.*. https://doi.org/10.1002/essoar.10501085.1

Lin, H., Huber, J. A., Gerl, G., & Hülsbergen, K.-J. (2016). Nitrogen balances and nitrogen-use efficiency of different organic and conventional farming systems. *Nutrient Cycling in Agroecosystems*, *105*(1), 1–23. https://doi.org/10.1007/s10705-016-9770-5

Lipman, J. G., & Conybeare, A. B. (1936). Preliminary note on the inventory and balance sheet of plant nutrients in the United States. Retrieved from http://agris.fao.org/agris-search/search.do?recordID=US201300618979

Liu, C., Watanabe, M., & Wang, Q. (2008). Changes in nitrogen budgets and nitrogen use efficiency in the agroecosystems of the Changjiang River basin between 1980 and 2000. *Nutrient Cycling in Agroecosystems*, *80*(1), 19–37. https://doi.org/10.1007/s10705-007-9118-2

Lorimor, J., Powers, W., & Sutton, A. (2004). Section 1: Manure characteristics. In *Manure management systems series* (pp. 2–24). Ames, Iowa: MidWest Plan Service. Retrieved from https://www.canr.msu.edu/uploads/files/ManureCharacteristicsMWPS-18_1.pdf

Lowrance, R. R., Leonard, R. A., Asmussen, L. E., & Todd, R. L. (1985). Nutrient budgets for agricultural watersheds in the southeastern coastal plain. *Ecology*, *66*(1), 287–296. https://doi.org/10.2307/1941330

Lu, C., & Tian, H. (2017). Global nitrogen and phosphorus fertilizer use for agriculture production in the past half century: Shifted hot spots and nutrient imbalance. *Earth System Science Data*, *9*, 181. https://doi.org/10.5194/essd-9-181-2017

Lu, C., Zhang, J., Cao, P., & Hatfield, J. L. (2019). Are we getting better in using nitrogen?: Variations in nitrogen use efficiency of two cereal crops across the United States. *Earth's Future*, *7*, 939–952. https://doi.org/10.1029/2019EF001155

Lun, F., Liu, J., Ciais, P., Nesme, T., Chang, J., Wang, R., et al. (2018). Global and regional phosphorus budgets in agricultural systems and their implications for phosphorus-use efficiency. *Earth System Science Data*, *10*(1), 1–18. https://doi.org/10.5194/essd-10-1-2018

Ma, L., Ma, W., Velthof, G., Wang, F., Qin, W., Zhang, F., & Oenema, O. (2010). Modeling nutrient flows in the food chain of China. *Journal of Environmental Quality*, *39*(4), 1279–1289. https://doi.org/10.2134/jeq2009.0403

Ma, L., Velthof, G., Wang, F., Qin, W., Zhang, W., Liu, Z., et al. (2012). Nitrogen and phosphorus use efficiencies and losses in the food chain in China at regional scales in 1980 and 2005. *Science of the Total Environment*, *434*, 51–61. https://doi.org/10.1016/j.scitotenv.2012.03.028

MacDonald, G. K., Bennett, E. M., Potter, P. A., & Ramankutty, N. (2011). Agronomic phosphorus imbalances across the world's croplands. *Proceedings of the National Academy of Sciences*, *108*(7), 3086–3091. https://doi.org/10.1073/pnas.1010808108

MacDonald, G. K., Bennett, E. M., & Taranu, Z. E. (2012). The influence of time, soil characteristics, and land-use history on soil phosphorus legacies: A global meta-analysis. *Global Change Biology*, *18*(6), 1904–1917. https://doi.org/10.1111/j.1365-2486.2012.02653.x

MacLeod, M. J., Vellinga, T., Opio, C., Falcucci, A., Tempio, G., Henderson, B., et al. (2017). Invited review: A position on the Global Livestock Environmental Assessment Model (GLEAM). *Animal*, *12*(2), 383–397. https://doi.org/10.1017/S1751731117001847

Malagó, A., Bouraoui, F., Pastori, M., & Gelati, E. (2019). Modelling nitrate reduction dtrategies from diffuse sources in the Po River Basin. *Water*, *11*, 1030. https://doi.org/10.3390/w11051030

McLellan, E. L., Cassman, K. G., Eagle, A. J., Woodbury, P. B., Sela, S., Tonitto, C., et al. (2018). The nitrogen balancing act: Tracking the environmental performance of food production. *Bioscience*, *68*(3), 194–203. https://doi.org/10.1093/biosci/bix164

Meisinger, J. J., Calderon, F., Jenkinson, D., Schepers, J., & Raun, W. (2008). Soil nitrogen budgets. *Agronomy*, *49*, 505. https://doi.org/10.2134/agronmonogr49.c13

Mekonnen, M. M., & Hoekstra, A. Y. (2018). Global anthropogenic phosphorus loads to freshwater and associated grey water footprints and water pollution levels: A high-resolution global study. *Water Resources Research*, *54*(1), 345–358. https://doi.org/10.1002/2017WR020448

Miyazaki, K., Eskes, H., Sudo, K., Boersma, K. F., Bowman, K., & Kanaya, Y. (2017). Decadal changes in global surface NO_x emissions from multi-constituent satellite data assimilation. *Atmospheric Chemistry and Physics*, *17*(2), 807–837. https://doi.org/10.5194/acp-2016-529

Mogollón, J., Beusen, A., Van Grinsven, H., Westhoek, H., & Bouwman, A. (2018). Future agricultural phosphorus demand according to the shared socioeconomic pathways. *Global Environmental Change*, *50*, 149–163. https://doi.org/10.1016/j.gloenvcha.2018.03.007

Mosier, A., Kroeze, C., Nevison, C., Oenema, O., Seitzinger, S., & Van Cleemput, O. (1998). Closing the global N_2O budget: Nitrous oxide emissions through the agricultural nitrogen cycle. *Nutrient Cycling in Agroecosystems*, *52*(2-3), 225–248. https://doi.org/10.1023/A:1009740530221

Mu, W., Van Middelaar, C., Bloemhof, J., Oenema, J., & De Boer, I. (2016). Nutrient balance at chain level: A valuable approach to benchmark nutrient losses of milk production systems. *Journal of Cleaner Production*, *112*, 2419–2428. https://doi.org/10.1016/j.jclepro.2015.09.116

Mueller, N. D., Lassaletta, L., Runck, B. C., Billen, G., Garnier, J., & Gerber, J. S. (2017). Declining spatial efficiency of global cropland nitrogen allocation. *Global Biogeochemical Cycles*, *31*, 245–257. https://doi.org/10.1002/2016GB005515

Mueller, N. D., West, P. C., Gerber, J. S., MacDonald, G. K., Polasky, S., & Foley, J. A. (2014). A tradeoff frontier for global nitrogen use and cereal production. *Environmental Research Letters*, *9*, 54002. https://doi.org/doi:10.1088/1748-9326/9/5/054002

NBSC. (2019). National Bureau of Statistics of China. Retrieved from http://data.stats.gov.cn

Nishina, K., Ito, A., Hanasaki, N., & Hayashi, S. (2017). Reconstruction of spatially detailed global map of NH^{4+} and NO^{3-}application in synthetic nitrogen fertilizer. *Earth System Science Data*, *9*(1). https://doi.org/10.5194/essd-9-149-2017

Norton, J. M., & Schimel, J. P. (2011). Nitrogen mineralization immobilization turnover. In *Handbook of Soil Science* (2nd ed., pp. 27.8–27.18). Boca Raton, FL: CRC Press.

Oenema, O. (2006). Nitrogen budgets and losses in livestock systems. Paper presented at the International Congress Series. https://doi.org/10.1016/j.ics.2006.02.040

Oenema, O., Kros, H., & de Vries, W. (2003). Approaches and uncertainties in nutrient budgets: Implications for nutrient management and environmental policies. *European Journal of Agronomy*, *20*(1), 3–16. https://doi.org/10.1016/s1161-0301(03)00067-4

Oita, A., Malik, A., Kanemoto, K., Geschke, A., Nishijima, S., & Lenzen, M. (2016). Substantial nitrogen pollution embedded in international trade. *Nature Geoscience*, *9*(2), 111. https://doi.org/10.1038/ngeo2635

Oliphant, A., Thenkabail, P. S., Teluguntla, P., Xiong, J., Gumma, M. K., Russell, G. C., & Kamini, Y. (2019). Mapping cropland extent of southeast and northeast Asia using multi-year time-series Landsat 30-m data using random forest classifier on Google Earth Engine. *International Journal of Applied Earth Observation and Geoinformation*, *81*, 110–124. https://doi.org/10.1016/j.jag.2018.11.014

Osmond, D. L., Hoag, D. L., Luloff, A. E., Meals, D. W., & Neas, K. (2015). Farmers' use of nutrient management: Lessons from watershed case studies. *Journal of Environmental Quality*, *44*(2), 382–390. https://doi.org/10.2134/jeq2014.02.0091

Patterson, P. H., Lorenz, E. S., Weaver, W. D., & Schwartz, J. H. (1998). Litter production and nutrients from commercial broiler chickens. *Journal of Applied Poultry Research*, *7*(3), 247–252.

Penuelas, J., Poulter, B., Sardans, J., Ciais, P., Van Der Velde, M., Bopp, L., et al. (2013). Human-induced nitrogen–phosphorus imbalances alter natural and managed ecosystems across the globe. *Nature Communications*, *4*(1), 2934. https://doi.org/10.1038/ncomms3934

Peoples, M., Brockwell, J., Herridge, D., Rochester, I., Alves, B., Urquiaga, S., et al. (2009). The contributions of nitrogen-fixing crop legumes to the productivity of agricultural systems. *Symbiosis*, *48*(1-3), 1–17. https://doi.org/10.1007/BF03179980

Perez, M. R. (2015). Regulating farmer nutrient management: A three-state case study on the Delmarva Peninsula. *Journal of Environmental Quality*, *44*(2), 402–414. https://doi.org/10.2134/jeq2014.07.0304

Pierrou, U. (1976). The global phosphorus cycle. *Ecological Bulletins*, *22*, 75–88. https://doi.org/10.1016/B0-08-043751-6/08153-6

Pinder, R. W., Davidson, E. A., Goodale, C. L., Greaver, T. L., Herrick, J. D., & Liu, L. (2012). Climate change impacts of US reactive nitrogen. *Proceedings of the National Academy of Sciences of the United States of America*, *109*(20), 7671–7675. https://doi.org/10.1073/pnas.1114243109

Powell, J. M., & Rotz, C. A. (2015). Measures of nitrogen use efficiency and nitrogen loss from dairy production systems. *Journal of Environmental Quality*, *44*(2), 336–344. https://doi.org/10.2134/jeq2014.07.0299

Powers, S. M., Bruulsema, T. W., Burt, T. P., Chan, N. I., Elser, J. J., Haygarth, P. M., et al. (2016). Long-term accumulation and transport of anthropogenic phosphorus in three river basins. *Nature Geoscience*, *9*(5), 353–356. https://doi.org/10.1038/ngeo2693

Prather, M. J., Hsu, J., DeLuca, N. M., Jackman, C. H., Oman, L. D., Douglass, A. R., et al. (2015). Measuring and modeling the lifetime of nitrous oxide including its variability. *Journal of Geophysical Research: Atmospheres*, *120*, 5693–5705. https://doi.org/10.1002/2015JD023267

Quemada, M., Lassaletta, L., Jensen, L. S., Godinot, O., Brentrup, F., Buckley, C., et al. (2020). Exploring nitrogen indicators of farm performance among farm types across several European case studies. *Agricultural Systems*, *177*, 102689. https://doi.org/10.1016/j.agsy.2019.102689

Ringeval, B., Augusto, L., Monod, H., van Apeldoorn, D., Bouwman, L., Yang, X., et al. (2017). Phosphorus in agricultural soils: Drivers of its distribution at the global scale. *Global Change Biology*, *23*(8), 3418–3432. https://doi.org/10.1111/gcb.13618

Robertson, G. P. (1986). Nitrogen: Regional contributions to the global cycle. *Environment: Science and Policy for Sustainable Development*, *28*(10), 16–29. https://doi.org/10.1080/00139157.1986.9928836

Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F. S. III, Lambin, E. F., et al. (2009). A safe operating space for humanity. *Nature*, *461*(7263), 472–475. https://doi.org/10.1038/461472a

Rowe, H., Withers, P. J., Baas, P., Chan, N. I., Doody, D., Holiman, J., et al. (2016). Integrating legacy soil phosphorus into sustainable nutrient management strategies for future food, bioenergy and water security. *Nutrient Cycling in Agroecosystems*, *104*(3), 393–412. https://doi.org/10.1007/s10705-015-9726-1

Roy, E. D., Richards, P. D., Martinelli, L. A., Della Coletta, L., Lins, S. R. M., Vazquez, F. F., et al. (2016). The phosphorus cost of agricultural intensification in the tropics. *Nature plants*, *2*(5), 16043. https://doi.org/10.1038/nplants.2016.43

Sachs, J. D., Schmidt-Traub, G., & Durand-Delacre, D. (2016). Preliminary sustainable development goal (SDG) index and dashboard. *Sustainable Development Solutions Network*, *15*.

Safford, H. D., Sawyer, S. C., Kocher, S. D., Hiers, J. K., & Cross, M. (2017). Linking knowledge to action: The role of boundary spanners in translating ecology. *Frontiers in Ecology and the Environment*, *15*(10), 560–568. https://doi.org/10.1002/fee.1731

Sattari, S., Van Ittersum, M., Giller, K., Zhang, F., & Bouwman, A. (2014). Key role of China and its agriculture in global sustainable phosphorus management. *Environmental Research Letters*, *9*(5), 054003. https://doi.org/10.1088/1748-9326/9/5/054003

Schindler, D. W., Hecky, R. E., Findlay, D. L., Stainton, M. P., Parker, B. R., Paterson, M. J., et al. (2008). Eutrophication of lakes cannot be controlled by reducing nitrogen input: Results of a 37-year whole-ecosystem experiment. *Proceedings of the National Academy of Sciences*, *105*(32), 11,254–11,258. http://www.pnas.org/content/105/32/11254.abstract, https://doi.org/10.1073/pnas.0805108105

Schlesinger, W. H. (2009). On the fate of anthropogenic nitrogen. *Proceedings of the National Academy of Sciences*, *106*(1), 203–208. https://doi.org/10.1073/pnas.0810193105

Schmidt-Traub, G., Kroll, C., Teksoz, K., Durand-Delacre, D., & Sachs, J. D. (2017). National baselines for the Sustainable Development Goals assessed in the SDG index and dashboards. *Nature Geoscience*, *10*(8), 547.

Scholz, R. W., & Hirth, T. (2015). Losses and efficiencies—From myths to data: Lessons learned from sustainable phosphorus management. *Resources, Conservation and Recycling*, *105*, 211–215. https://doi.org/10.1016/j.resconrec.2015.10.005

Schroeck, A. M., Gaube, V., Haas, E., & Winiwarter, W. (2019). Estimating nitrogen flows of agricultural soils at a landscape level—A modelling study of the Upper Enns Valley, a long-term socio-ecological research region in Austria. *Science of the Total Environment*, *665*, 275–289. https://doi.org/10.1016/j.scitotenv.2019.02.071

Schwede, D. B., & Lear, G. G. (2014). A novel hybrid approach for estimating total deposition in the United States. *Atmospheric Environment*, *92*, 207–220. https://doi.org/10.1016/j.atmosenv.2014.04.008

Seitzinger, S. P., Harrison, J. A., Dumont, E., Beusen, A. H. W., & Bouwman, A. F. (2005). Sources and delivery of carbon, nitrogen, and phosphorus to the coastal zone: An overview of Global Nutrient Export from Watersheds (NEWS) models and their application. *Global Biogeochemical Cycles*, *19*, GB4S01. https://doi.org/10.1029/2005GB002606

Sela, S., Woodbury, P. B., Marjerison, R., & van Es, H. (2019). Towards applying N balance as a sustainability indicator for the US cornbelt: Realistic achievable targets, spatio-temporal variability and policy implications. *Environmental Research Letters*. https://doi.org/10.1088/1748-9326/ab1219

Sheldrick, W., Syers, J. K., & Lingard, J. (2003). Contribution of livestock excreta to nutrient balances. *Nutrient Cycling in Agroecosystems*, *66*(2), 119–131.

Sinha, E., & Michalak, A. M. (2016). Precipitation dominates interannual variability of riverine nitrogen loading across the continental United States. *Environmental Science & Technology*, *50*(23), 12,874–12,884. https://doi.org/10.1021/acs.est.6b04455

Smith, K. A., Mosier, A. R., Crutzen, P. J., & Winiwarter, W. (2012). The role of N_2O derived from crop-based biofuels, and from agriculture in general, in Earth's climate. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *367*(1593), 1169–1174. https://doi.org/10.1098/rstb.2011.0313

Snyder, G. H. (1996). Nitrogen losses by leaching and runoff: Methods and conclusions. In *Nitrogen Economy in Tropical Soils* (pp. 427–432). Dordrecht, Netherlands: Springer. https://doi.org/10.1007/978-94-009-1706-4_41

Söderlund, R., & Svensson, B. H. (1976). The global nitrogen cycle. *Ecological Bulletins*, *22*, 23–73. https://doi.org/10.1002/9781118280874.ch4

Springmann, M., Clark, M., Mason-D'Croz, D., Wiebe, K., Bodirsky, B. L., Lassaletta, L., et al. (2018). Options for keeping the food system within environmental limits. *Nature*, *562*(7728), 519–525. https://doi.org/10.1038/s41586-018-0594-0

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., et al. (2015). Planetary boundaries: Guiding human development on a changing planet. *Science*, *348*(6240), 1217. https://doi.org/10.1126/science.aaa9629

Sullivan, B. W., Smith, W. K., Townsend, A. R., Nasto, M. K., Reed, S. C., Chazdon, R. L., & Cleveland, C. C. (2014). Spatially robust estimates of biological nitrogen (N) fixation imply substantial human alteration of the tropical N cycle. *Proceedings of the National Academy of Sciences*, *111*(22), 8101–8106. https://doi.org/10.1073/pnas.1320646111

Sutton, M. A., Bleeker, A., Howard, C., Bekunda, M., Grizzetti, B., de Vries, W., et al. (2013). *Our nutrient world. The challenge to produce more food & energy with less pollution*. Edinburgh: NERC/Centre for Ecology & Hydrology Retrieved from https://library.wur.nl/WebQuery/wurpubs/reports/434951

Svensson, B. H., & Söderlund, R. (1976). Editors' Preface. *Ecological Bulletins*, *22*, 7–8. Retrieved from http://www.jstor.org/stable/20112517

Swaney, D. P., Hong, B., Ti, C., Howarth, R. W., & Humborg, C. (2012). Net anthropogenic nitrogen inputs to watersheds and riverine N export to coastal waters: A brief overview. *Current Opinion in Environmental Sustainability*, *4*(2), 203–211. https://doi.org/10.1016/j.cosust.2012.03.004

Tenorio, F. A., Eagle, A. J., McLellan, E. L., Cassman, K. G., Howard, R., Below, F. E., et al. (2019). Assessing variation in maize grain nitrogen concentration and its implications for estimating nitrogen balance in the US north central region. *Field Crops Research*, *240*, 185–193. https://doi.org/10.1016/j.fcr.2018.10.017

Thompson, R. L., Lassaletta, L., Patra, P. K., Wilson, C., Wells, K. C., Gressent, A., et al. (2019). Acceleration of global N_2O emissions seen from two decades of atmospheric inversion. *Nature Climate Change*, *9*(12), 993–998. https://doi.org/10.1038/s41558-019-0613-7

Tian, H., Banger, K., Bo, T., & Dadhwal, V. K. (2014). History of land use in India during 1880–2010: Large-scale land transformations reconstructed from satellite data and historical archives. *Global and Planetary Change*, *121*, 78–88. https://doi.org/10.1016/j.gloplacha.2014.07.005

Tian, H., Yang, J., Xu, R., Lu, C., Canadell, J. G., Davidson, E. A., et al. (2019). Global soil nitrous oxide emissions since the preindustrial era estimated by an ensemble of terrestrial biosphere models: Magnitude, attribution, and uncertainty. *Global Change Biology*, *25*(2), 640–659. https://doi.org/10.1111/gcb.14514

UNEP. (2013). Drawing down N_2O to protect climate and the ozonelayer: A UNEP synthesis report. Nairobi, Kenya: United Nations Environment Programme (UNEP). Retrieved from https://ueaeprints.uea.ac.uk/66211/1/UNEPN2Oreport.pdf

Unkovich, M., Herridge, D., Peoples, M., Cadisch, G., Boddey, B., Giller, K., et al. (2008). Measuring plant-associated nitrogen fixation in agricultural systems. In (pp. 258): Australian Centre for International Agricultural Research (ACIAR).

USDA. (2013). USDA Nutrient Database for Standard Reference, Release 26. Retrieved from https://www.ars.usda.gov/northeast-area/beltsville-md-bhnrc/beltsville-human-nutrition-research-center/nutrient-data-laboratory/docs/sr26-home-page/

USGS. (2019). Phosphate Rock Statistics and Information. Retrieved from https://www.usgs.gov/centers/nmic/phosphate-rock-statistics-and-information

Uwizeye, A., Gerber, P. J., Opio, C. I., Tempio, G., Mottet, A., Makkar, H. P., et al. (2019). Nitrogen flows in global pork supply chains and potential improvement from feeding swill to pigs. *Resources, Conservation and Recycling*, *146*, 168–179. https://doi.org/10.1016/j.resconrec.2019.03.032

Uwizeye, A., Gerber, P. J., Schulte, R. P., & de Boer, I. J. (2016). A comprehensive framework to assess the sustainability of nutrient use in global livestock supply chains. *Journal of Cleaner Production*, *129*, 647–658. https://doi.org/10.1016/j.jclepro.2016.03.108

Van Damme, M., Clarisse, L., Whitburn, S., Hadji-Lazaro, J., Hurtmans, D., Clerbaux, C., & Coheur, P.-F. (2018). Industrial and agricultural ammonia point sources exposed. *Nature*, *564*(7734), 99–103. https://doi.org/10.1038/s41586-018-0747-1

Van der Hoek, K. W. (1998). Nitrogen efficiency in global animal production. *Environmental Pollution*, *102*, 127–132. https://doi.org/10.1016/B978-0-08-043201-4.50023-X

Van der Pol, F. (1992). Soil mining: An unseen contributor to farm income in southern Mali: Intermediate technology. Retrieved from http://edepot.wur.nl/419522#page=74

van Groenigen, J. W., van Kessel, C., Hungate, B. A., Oenema, O., Powlson, D. S., & van Groenigen, K. J. (2017). Sequestering soil organic carbon: A nitrogen dilemma. *Environmental Science & Technology*, *51*(9), 4738–4739. https://doi.org/10.1021/acs.est.7b01427

Van Kauwenbergh, S. J. (2010). World phosphate rock reserves and resources: IFDC muscle shoals. Retrieved from http://www.firt.org/sites/default/files/SteveVanKauwenbergh_World_Phosphate_Rock_Reserve.pdf

Van Meter, K., Van Cappellen, P., & Basu, N. (2018). Legacy nitrogen may prevent achievement of water quality goals in the Gulf of Mexico. *Science*, *360*(6387), 427–430. https://doi.org/10.1126/science.aar4462

Van Meter, K. J., Basu, N. B., Veenstra, J. J., & Burras, C. L. (2016). The nitrogen legacy: Emerging evidence of nitrogen accumulation in anthropogenic landscapes. *Environmental Research Letters*, *11*(3), 035014. https://doi.org/10.1088/1748-9326/11/3/035014

Viets, F. G. Jr. (1971). Water quality in relation to farm use of fertilizer. *Bioscience*, *21*(10), 460–467. https://doi.org/10.2307/1295787

Vitousek, P. M., Aber, J. D., Howarth, R. W., Likens, G. E., Matson, P. A., Schindler, D. W., et al. (1997). Human alteration of the global nitrogen cycle: Sources and consequences. *Ecological Applications*, *7*(3), 737–750. https://doi.org/10.1890/1051-0761(1997)007[0737:HAOTGN]2.0.CO;2

Vitousek, P. M., Menge, D. N., Reed, S. C., & Cleveland, C. C. (2013). Biological nitrogen fixation: Rates, patterns and ecological controls in terrestrial ecosystems. *Philosophical Transactions of the Royal Society, B: Biological Sciences*, *368*(1621), 20130119. https://doi.org/10.1098/rstb.2013.0119

Vitousek, P. M., Naylor, R., Crews, T., David, M. B., Drinkwater, L., Holland, E., et al. (2009). Nutrient imbalances in agricultural development. *Science*, *324*(5934), 1519–1520. https://doi.org/10.1126/science.1170261

Wall, T. U., McNie, E., & Garfin, G. M. (2017). Use-inspired science: making science usable by and useful to decision makers. *Frontiers in Ecology and the Environment*, *15*(10), 551–559. https://doi.org/10.1002/fee.1735

Wang, Y., Law, R. M., & Pak, B. (2009). A global model of carbon, nitrogen and phosphorus cycles for the terrestrial biosphere. *Biogeosciences*, *7*, 2261–2282. https://doi.org/10.5194/bg-7-2261-2010

Westhoek, H., Lesschen, J. P., Leip, A., Rood, T., Wagner, S., De Marco, A., et al. (2014). Food choices, health and environment: Effects of cutting Europe's meat and dairy intake. *Global Environmental Change*, *26*, 196–205. https://doi.org/10.1016/j.gloenvcha.2014.02.004

Xu, R., Tian, H., Pan, S., Dangal, S. R., Chen, J., Chang, J., et al. (2019). Increased nitrogen enrichment and shifted patterns in the world's grassland: 1860–2016. *Earth System Science Data*, *11*(1), 175–187. https://doi.org/10.5194/essd-11-175-2019

Xu, W., Luo, X., Pan, Y., Zhang, L., Tang, A., Shen, J., et al. (2015). Quantifying atmospheric nitrogen deposition through a nationwide monitoring network across China. *Atmospheric Chemistry and Physics*, *15*(21), 12,345–12,360. https://doi.org/10.5194/acp-15-12345-2015

Xue, X., & Landis, A. E. (2010). Eutrophication potential of food consumption patterns. *Environmental Science & Technology*, *44*(16), 6450–6456. https://doi.org/10.1021/es9034478

Yan, X., Ti, C., Vitousek, P., Chen, D., Leip, A., Cai, Z., & Zhu, Z. (2014). Fertilizer nitrogen recovery efficiencies in crop production systems of China with and without consideration of the residual effect of nitrogen. *Environmental Research Letters*, *9*(9), 095002. https://doi.org/10.1088/1748-9326/9/9/095002

Yang, X., Post, W. M., Thornton, P. E., & Jain, A. (2013). The distribution of soil phosphorus for global biogeochemical modeling. *Biogeosciences*, *10*(4), 2525–2537. https://doi.org/10.5194/bgd-9-16347-2012

Yu, G., Jia, Y., He, N., Zhu, J., Chen, Z., Wang, Q., et al. (2019). Stabilization of atmospheric nitrogen deposition in China over the past decade. *Nature Geoscience*, *12*(6), 424–429. https://doi.org/10.1038/s41561-019-0352-4

Zebarth, B. J., Hii, B., Liebscher, H., Chipperfield, K., Paul, J. W., Grove, G., & Szeto, S. Y. (1998). Agricultural land use practices and nitrate contamination in the Abbotsford Aquifer, British Columbia, Canada. *Agriculture, Ecosystems & Environment*, *69*(2), 99–112.

Zhang, C., Ju, X., Powlson, D. S., Oenema, O., & Smith, P. (2019). Nitrogen surplus benchmarks for controlling N pollution in the main cropping systems of China. *Environmental Science & Technology*. https://doi.org/10.1021/acs.est.8b06383

Zhang, C., Liu, S., Wu, S., Jin, S., Reis, S., Liu, H., & Gu, B. (2019). Rebuilding the linkage between livestock and cropland to mitigate agricultural pollution in China. *Resources, Conservation and Recycling*, *144*, 65–73. https://doi.org/10.1016/j.resconrec.2019.01.011

Zhang, J., Beusen, A. H. W., Van Apeldoorn, D. F., Mogollón, J. M., Yu, C., & Bouwman, A. F. (2017). Spatiotemporal dynamics of soil phosphorus and crop uptake in global cropland during the 20th century. *Biogeosciences*, *14*(8), 2055–2068. https://doi.org/10.5194/bg-14-2055-2017

Zhang, L., Chen, Y., Zhao, Y., Henze, D., Zhu, L., Song, Y., et al. (2018). Agricultural ammonia emissions in China: reconciling bottom-up and top-down estimates. *Atmospheric Chemistry and Physics*, *18*(1), 339–355. https://doi.org/10.5194/acp-18-339-2018

Zhang, X. (2017). Biogeochemistry: A plan for efficient use of nitrogen fertilizers. *Nature*, *543*(7645), 322–323. https://doi.org/10.1038/543322a

Zhang, X., & Davidson, E. A. (2019). Sustainable nitrogen management index. *Earth and Space Science Open Archive.*. https://doi.org/10.1002/essoar.10501111.1

Zhang, X., Davidson, E. A., Mauzerall, D. L., Searchinger, T. D., Dumas, P., & Shen, Y. (2015). Managing nitrogen for sustainable development. *Nature*, *528*(7580), 51–59. https://doi.org/10.1038/nature15743

Zhang, X., Lee, X., Griffis, T., Baker, J., & Xiao, W. (2014). Estimating greenhouse gas fluxes from an agriculture-dominated landscape using multiple planetary boundary layer methods. *Atmospheric Chemistry and Physics*, *14*(3), 10,705–10,719. https://doi.org/10.5194/acpd-14-3231-2014

Zhang, X., Lee, X., Griffis, T. J., Andrews, A. E., Baker, J. M., Erickson, M. D., et al. (2015). Quantifying nitrous oxide fluxes on multiple spatial scales in the Upper Midwest, USA. *International Journal of Biometeorology*, *59*(3), 299–310. https://doi.org/10.1007/s00484-014-0842-4

Zhang, X., Mauzerall, D. L., Davidson, E. A., Kanter, D. R., & Cai, R. (2015). The economic and environmental consequences of implementing nitrogen-efficient technologies and management practices in agriculture. *Journal of Environmental Quality*, *44*(2), 312–324. https://doi.org/10.2134/jeq2014.03.0129