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FEATURE

Plant nutrient management and risks of nitrous oxide emission

Clifford S. Snyder and Paul E. Fixen

Global fertilizer production and farmer use has made it possible to provide 40% to 60% of the global crop and food production necessary to sustain the human family (Erisman et al. 2008; Stewart et al. 2005). Because the world's population is growing, it has been projected that we may need to produce 50% more food by 2050 to meet the needs of nine billion people (Tomlinson 2011; Whitacre et al. 2010). Nitrogen (N) fertilizers have had a tremendously beneficial impact on society, and at least half of humanity currently depends on the food production made possible by the Haber–Bosch ammonia (NH_3) synthesis process (Erisman et al. 2008). The agricultural community and nonagrarian society have a heightened awareness and are recognizing that it is becoming more important to provide balance in the use of N and other nutrient inputs for the production of food, feed, fiber, and biofuel to help achieve sustainability goals (IFA 2007). The environmental risks and consequences of increased reactive N inputs to the earth's ecosystems are also being considered and have been the subject of many environmental reports (Davidson et al. 2012; Galloway et al. 2003, 2004; Follett and Delgado 2002; Howarth 2010; Matson 1998; Schlesinger et al. 2006; Seitzinger et al. 2010; Vitousek 1997). Interrelated issues of population growth, sustainable food production, climate change, and environmental protection have attracted the interests of many diverse groups and were, for example, the subject of a scholarly symposium entitled, Estimating Earth's Human-Carrying Capacity, at the 2011 American Association for the Advancement of Science meetings in Washington, DC (AAAS 2011).

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FERTILIZER NITROGEN USE AND NITROUS OXIDE EMISSIONS

The three greenhouse gases (GHGs) associated with agriculture—carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O)—have different heat trapping and atmospheric turnover rates. Based on a 100-year timeframe, the unit masses of CH_4 and N_2O are considered to have 23 and 296 times the global warming potential, respectively, as a unit of CO_2 (IPCC 2001).

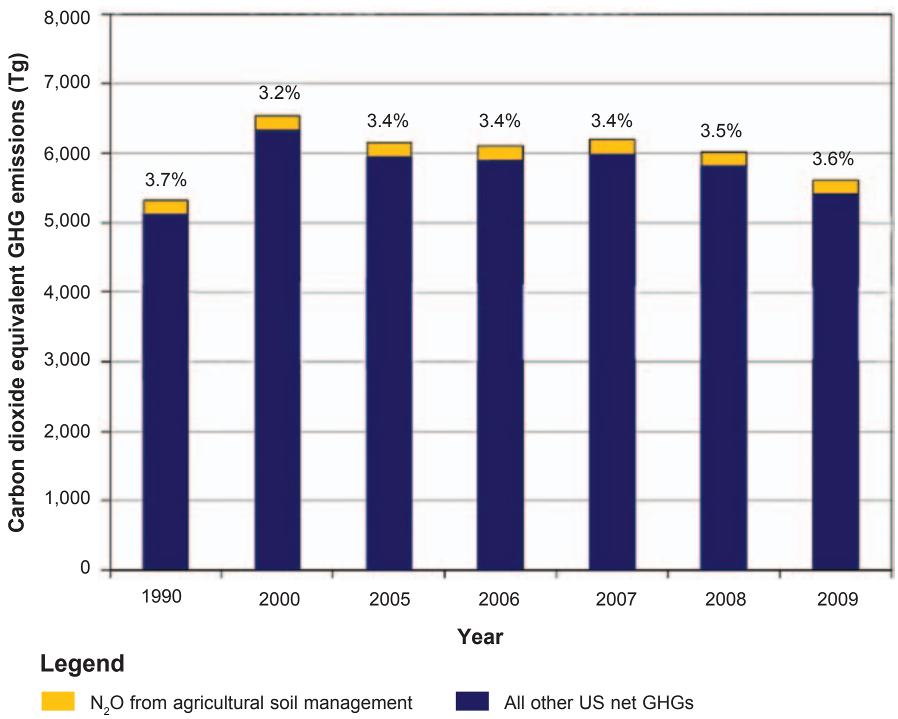
In comparison to CH_4 and N_2O, CO_2 is cycled through agricultural cropping systems in the largest amounts. Plants release oxygen (O_2) as they consume large amounts of CO_2 through photosynthesis, and all plant products eventually convert back to CO_2 when consumed or when they decompose. The net emission of CO_2 is small in comparison to its total cycling in agriculture and is mostly due to energy use on farm and in the manufacture and transport of agricultural products (Flynn and Smith 2010; Snyder et al. 2009). The CO_2 concentrations in the atmosphere have increased from approximately 280 parts per million (ppm) in preindustrial times to >390 ppm in 2010 (ASA CSSA SSSA 2010; NOAA 2011). The N_2O concentration in the atmosphere increased from 270 parts per billion (ppb) from preindustrial times to 319 ppb in 2005, an increase of about 0.26% per year. Based on a modeling approach, the majority of this increased N_2O concentration has been attributed to manure and fertilizer N inputs (Davidson 2009). However, agricultural and urban fertilizer N consumption accounted for only 31% of all N inputs in the United States in 2002 (USEPA 2011a).

The total global fertilizer N consumption in 1970 was about 31 million t (34 million tn), 82 million t (90 million tn) in 2000, and exceeded 102 million t (112 million tn) in 2009. The US fertilizer N consumption represented 23%, 13%, and 11% of the world total during these respective years (IFA 2011). Although as much as 60% of the global total N_2O emissions and 50% of the global CH_4 emissions have been attributed to agriculture, world fertilizer N consumption may account for less than 7% to 10% of the global total GHG emissions (USEPA 2006, 2011b). In the United States, agricultural soil management, which includes fertilizer N use, represented 69% of the US total N_2O emissions but accounted for less than 4% of the US total CO_2-equivalent (CO_2e) GHG emissions, and this percentage has not risen since 1990 (USEPA 2011c) (figure 1). Land use change, associated with the clearing of forests and the conversion of native lands for agricultural production, accounted for between 6% and 17% of the global total GHG emissions (Foley et al. 2005; Flynn and Smith 2010). Land use change—from natural forests, grasslands, and wetlands to more intensive uses—represents perhaps the biggest threat to global GHG emission increases, especially since roughly 35% of anthropogenic CO_2 emissions since 1850 have resulted directly from land use change (Foley et al. 2005). While it can be argued that crop agriculture has many opportunities to reduce its GHG emissions footprint, it is clear that things would be much worse if not for the global adoption of modern cropping technologies, including fertilizer use (Burney et al. 2010). Global pressures to minimize CO_2e emissions have increased in order to reduce anthropogenic impacts on global warming and climate change. Because of N_2O's large CO_2e and the comparative mitigation offset per dollar invested, it is not surprising that N_2O emission reduction protocols have been adopted in Alberta, Canada, and are under development in other parts of Canada and the United States as part of voluntary offset programs (C-AGG 2010).

Emissions of N_2O "induced" by fertilizer N in most major cropping systems have been reported to range from less than 0.5% to more than 2%, with an estimated global average of 0.9% (Bouwman et al. 2002a, 2002b; Stehfest and Bouwman 2006). Burney et al. (2010) found that even with greater global GHG emissions

Figure 1

Net greenhouse gas (GHG) emissions and nitrous oxide (N_2O) associated with agricultural soil management in the United States. Agricultural soil management includes fertilizer application and cropping practices, accounting for 69% of all US N_2O emissions in 2009. (USEPA 2011c)



associated with increased fertilizer production and application, the net effect of the associated increased crop production since 1961 has been the avoidance of up to 590 Gt (650 billion tn) of CO_2e or 161 Gt (177 billion tn) of carbon(C), an amount equal to roughly 100 years of the current annual total GHG emissions from the United States (USEPA 2011c).

IMPROVING NITROGEN USE EFFICIENCY AND EFFECTIVENESS

One of the most pressing challenges in meeting these societal demands is the improvement of crop N use effectiveness and efficiency since the apparent above-ground, growing season recovery of applied N by corn (*Zea mays* L.), for example, typically ranges below 40% to 50% (Dobermann 2007; Ladha et al. 2005; Randall et al. 2008). Apparent N recoveries above 70% may be achieved for many cereal crops by using intensive site-specific nutrient management (Dobermann and Cassman 2002), based on the principles of 4R Nutrient Stewardship (right source at the right rate, time, and place) (Bruulsema et al. 2008), when used in tandem with optimum management of other cropping system resources and inputs. The US Environmental Protection Agency (2011a) has advocated a 25% increase in crop N use efficiency above current levels, and such improvements seem reasonable and within reach on many farms (Dobermann 2007; Kitchen and Goulding 2001) through more intensive, skilled nutrient and cropping system management. Poorly managed, imbalanced, and inefficient agricultural N use impairs the ability to provide food, feed, fiber and biofuel; raises the risks for N loss to groundwater and surface water resources; and increases the potential for direct and indirect emissions of the potent GHG, N_2O (Baker and Johnson 1981; Cassman et al. 2002; USEPA 2011a, David et al. 2010; Delgado 2002; Delgado et al. 2005; Delgado and Follett 2010; Dobermann 2007; Dobermann and Cassman 2002; Follett and Delgado 2002; Hatfield et al. 2009; Mosier et al. 2002; Randall and Goss 2001; Roberts et al. 2010; Snyder et al. 2009; Van Groenigen et al. 2011).

In crop agriculture, N_2O emissions can have a large influence on the GHG budget associated with different cropping systems on individual fields (Robertson et al. 2000). Yet, Adviento-Borbe et al. (2007) found that net GHG emissions from corn–corn and corn–soybean (*Glycine max* L.) systems can be kept low when management is optimized to achieve the yield potential. The optimization of crop yields on existing agricultural lands while protecting and preserving wetlands, forests, and grasslands—termed ecological intensification (Cassman 1999)—may help reduce the risks of increased GHG emissions from crop agriculture (Burney et al. 2010) because it "creates large sinks for CO_2 and mineral N, thereby providing the prerequisite for sequestering atmospheric CO_2 and avoiding large N_2O emissions that could result from inefficient utilization of soil or fertilizer N" (Adviento-Borbe et al. 2007).

Nutrient management plays a large role in helping optimize crop response to inputs (Bruulsema et al. 2009; Cassman et al. 2002; Dobermann and Cassman 2002; Dobermann 2007) and should be included as a component of the overall cropping system management plan, which includes appropriate soil and water conservation practices (Delgado and Bausch 2005), integrated pest management, and the use of adapted crop varieties and hybrids (which are input-efficient and responsive to management [Cassman 1999]) at optimum densities (Ping et al. 2008). According to Adviento-Borbe et al. (2007), "Intensification of cropping does not necessarily increase GHG emissions and GWP [global warming potential] of agricultural systems provided that crops are grown with best management practices and near yield potential levels, resulting in high resource use efficiency." The successful implementation of agricultural GHG mitigation, in synergy with sustainability goals, may depend to a considerable extent on the price of CO_2e.

Although there is significant mitigation potential, many barriers must be overcome with appropriate policies, education, and societal understanding (C-AGG 2010;

Smith et al. 2007, 2008). A special issue of the *Journal and Soil Water Conservation* (Volume 63, Number 6) was devoted in 2008 to the benefits, accomplishments, and challenges—including some nutrient management challenges—during the first five years of the USDA Conservation Effects Assessment Project (CEAP). Recent USDA reports have indicated there is significant opportunity to improve management of N and phosphorus (P) to improve crop production and to help mitigate impacts on water quality. For example, the Upper Mississippi River Basin CEAP report stated that complete and consistent use of nutrient management practices is generally lacking and that 62% of the acres in the upper Mississippi River Basin need additional management intervention to reduce the loss of N or P from farm fields (NRCS 2010), while the CEAP report for the Chesapeake Bay indicated that only 9% of the acres fully meet USDA Natural Resources Conservation Service nutrient rate, timing, and method of application nutrient management criteria (NRCS 2011). Improved management to increase crop N uptake and applied N recovery is likely to reduce direct N_2O emissions as well as the indirect N_2O emissions that may be associated with other N loss pathways (e.g., nitrate [NO_3] leaching, drainage, and runoff) (Eagle et al. 2010; USEPA 2011c; Millar et al. 2010; Snyder et al. 2009).

The importance of improved N use efficiency in meeting global food demands while protecting natural resources and environmental quality has been emphasized in several reports (ASA CSSA SSSA 2010; Dobermann and Cassman 2002; Cassman et al. 2003; Snyder and Bruulsema 2007). These reports illustrated the significant yield gap that exists between farmer average yields and yields achieved under more intensive site-specific best management practices. Some examples of fertilizer N best management practices to improve crop yields, increase N use efficiency and effectiveness, and reduce N_2O emissions were outlined by Snyder (2008) and Snyder et al. (2009). While some reports (Millar et al. 2010) argue that data on fertilizer N source, timing, and place of application are insufficient to warrant N_2O emission reduction protocols that include any N management factors beyond simple consideration of reduced N application rates, there is a considerable and growing body of evidence that indicates otherwise. The papers by the following authors report N_2O emission reductions with changes in N source, time, and place of application, ranging from 20% to more than 80%: Akiyama et al. (2010); Archer and Halvorson (2010); Breitenbeck and Bremner (1986a, 1986b); Bronson et al. (1992); Bruulsema et al. (2011); Burton et al. (2008); Delgado and Mosier (1996); Drury et al. (2006); Halvorson et al. (2010); Hyatt et al. (2010); Kahlil et al. (2009); Liu et al. (2006); Magiatto et al. (2000); Motavalli et al. (2008); Nyakatawa et al. (2011); Pathak and Nedwell (2001); Sehy et al. (2003); Shoji et al. (2001); Tenuta and Beauchamp (2003); Venterea et al. (2010, 2011); and Zebarth et al. (2008b). These and other reports on reductions in N_2O emissions, which are possible with site-specific management of N source, time, and place of application, are especially important in view of the larger objective of increasing crop yields for a burgeoning population. There is some concern among practicing agronomists that a N_2O emission reduction effort that is too narrowly focused on N rate reduction alone could jeopardize opportunities to increase crop production needs of the global population, limit CO_2 capture by skillfully managed crops, and interfere with restoration and maintenance of soil organic matter (Ladha et al. 2011). Further, Six et al. (2002) reported, "...from a global change standpoint, agricultural management practices that may reduce N_2O-fluxes (e.g., application time and method of N fertilizer, precision farming techniques, alternative crop rotations, type of cover crop, addition of nitrification inhibitors) should be investigated and implemented if found to be feasible."

REDUCING RESIDUAL SOIL NITRATE TO MINIMIZE POTENTIAL LOSSES

Research has shown that N_2O emissions may occur during nitrification, via related biochemical processes under aerobic conditions, as well as during denitrification via less oxygenated conditions within the N cycle (Coyne 2008; Freney et al. 1979; Parton et al. 1988; Venterea 2007). Yet, the magnitude of direct N_2O-N loss from agricultural fields—generally less than 2 to 8 kg N ha^{-1} y^{-1} (1.8 to 7 lb N ac^{-1} yr^{-1})—is often less than the quantity of N that may be lost via NO_3 leaching, drainage, and runoff or via NH_3 volatilization. Crutzen et al. (2008) have suggested that indirect emissions of N may range from three to five times direct emissions. Losses of nitrate-nitrogen (NO_3-N) via subsurface drainage in the United States and Europe may exceed 30 to 35 kg N ha^{-1} y^{-1} (27 to 31 lb N ac^{-1} yr^{-1}), even in systems that have not been cropped, fertilized, or cultivated for over 25 years (Randall et al. 2010). Ammonia loss from applied urea or urea-containing N sources is usually relatively small but may range between 10% and 40% of the applied N and can approach 100% if not properly soil incorporated (Mikkelsen 2009). If one assumes an average N application rate of 100 kg N ha^{-1} (89 lb N ac^{-1}) and considers that urea and urea-containing fertilizers may constitute roughly half of the fertilizer N consumption, a crude estimate of NH_3 emissions from land receiving fertilizer N might range from 5 to 20 kg N ha^{-1} (4 to 18 lb N ac^{-1}). However, it is important to note that NH_3 emissions from livestock and poultry are a much larger source of NH_3 (71% of US total) than emissions from fertilizer use (14% of US total) (USEPA 2004; Bittman and Mikkelsen 2009). Direct (and indirect) N_2O emissions generally increase with increasing N inputs but usually do not increase markedly (nonlinearly) until the applied N inputs appreciably exceed crop N uptake (Bouwman et al. 2002; McSwiney and Robertson 2005; Zebarth et al. 2008a). For that reason, Snyder et al. (2009) and van Groenigen et al. (2010) have stated that evaluation of N_2O emissions per unit land area (area-scaled) are important, but they have asserted that it is equally important to express and evaluate N_2O emissions per unit of crop production or on a yield-scaled basis (Venterea et al. 2011).

Direct N_2O emissions have been related to soil NO_3 intensity, the calculated cumulative value of daily soil NO_3 concentrations in the surface soil (Zebarth et al. 2008b). Because of this observation and the indirect N_2O emissions that may result from NO_3-N losses to water resources (Beaulieu et al. 2011), it is considered important to manage N and other inputs to enhance crop recovery and to do so in a manner that limits the potential for sizeable NO_3-N accumulation in the soil profile. Providing balanced plant nutrition is a key way to help minimize the potential for soil profile NO_3-N accumulation. For example, long-term corn nutrition research in Kansas has shown the importance of proper P nutrition in improving crop N recovery and reducing soil profile NO_3-N accumulation (figure 2), and research with corn in Ohio has shown that adequate potassium (K) nutrition can improve crop N recovery substantially (figure 3). These data and results from numerous other studies help underscore the need to provide other essential plant nutrients in balance with optimum N rates to achieve crop yield and N recovery goals.

US SOIL FERTILITY STATUS AND NITROGEN BALANCE ESTIMATES

In 2010, the International Plant Nutrition Institute (IPNI) developed a summary report on its survey of public and private soil testing laboratories in North America to determine the P, K, pH, magnesium (Mg), sulfur (S), zinc (Zn), and chloride (Cl) fertility status. The soil testing summary involved approximately 4.4 million soil samples for the 2010 crop year (IPNI 2010b) and was the most comprehensive soil fertility evaluation ever conducted in North America. The median soil P was 25 ppm and represented a 6 ppm decline since the IPNI 2005 soil test summary, with approximately 42% of the samples testing below 20 ppm, a level considered an agronomic optimum or critical level in many states. The median soil K level declined 4 ppm below the 2005 levels to 150 ppm, with approximately 34% of the soil samples testing below 120 ppm, a level considered an agronomic optimum or critical level in the majority of states. These recent soil fertility summary data indicate that a large portion of crop land in the United States has soil test P and K levels below agronomic and economically optimum levels. A comparison of these soil P and K summary results with data shown in figures 1 and 2 indicates that crop N recovery may be significantly impaired by inadequate soil P and K fertility on many hectares in the United States. The comparison also indicates that impaired crop N recovery efficiency may be contribut-

Figure 2

Adequate phosphorus (P) fertilization of corn reduces soil profile nitrate-nitrogen (NO_3-N) and the potential for nitrate leaching (Schlegel et al. 1996).

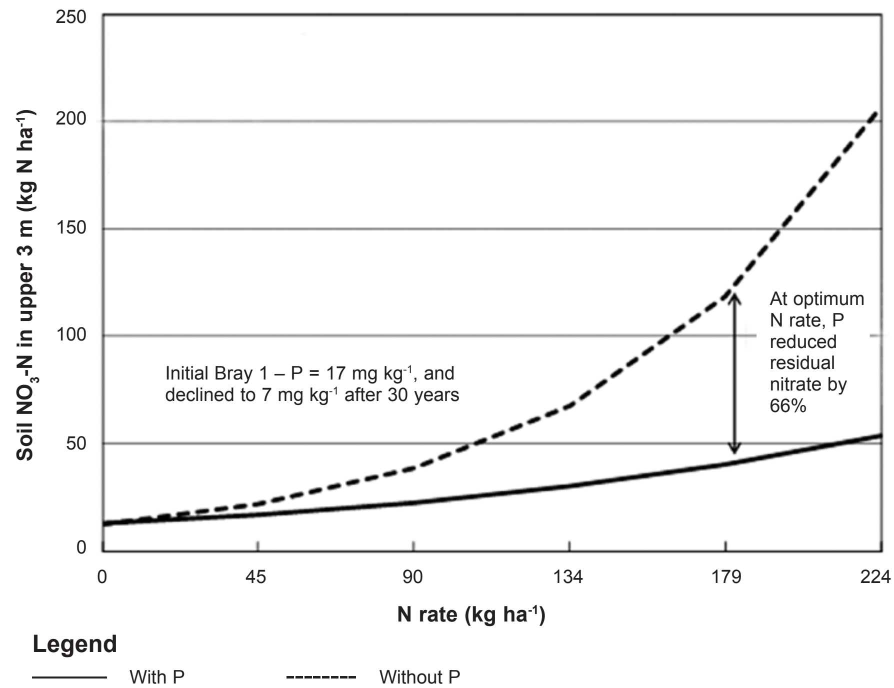
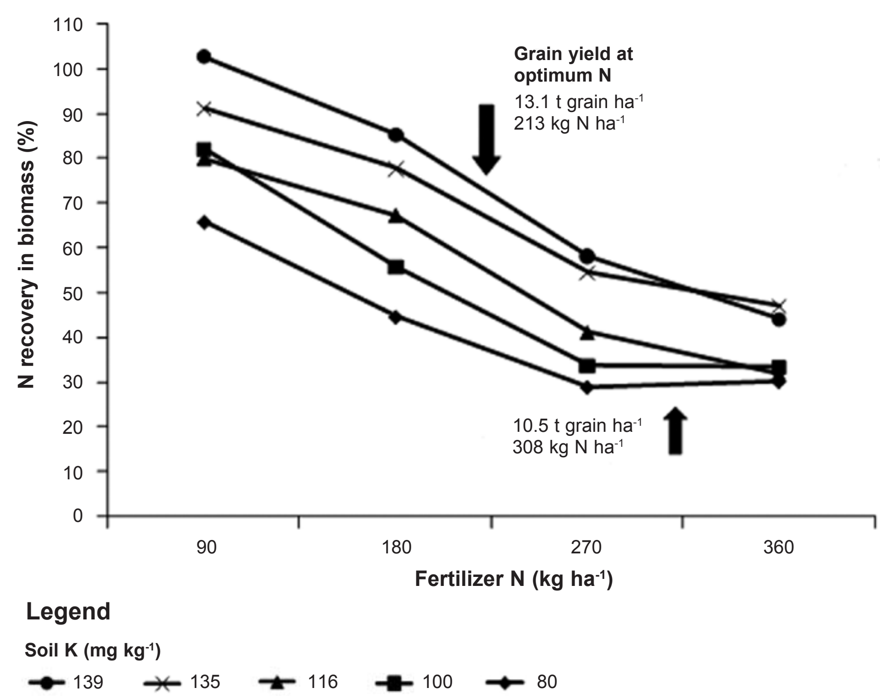


Figure 3

Adequate potassium (K) fertilization increases apparent nitrogen (N) recovery by corn (Johnson et al. 1997).



ing to soil profile NO_3-N accumulation and increased risks of NO_3-N leaching and drainage losses, potentially resulting in risks of elevated indirect N_2O emissions.

Estimates of current N balance on croplands in the United States can provide insight on production and environmental consequences of cropping system and nutrient management. In recognition of the interaction of N and other nutrients in crops, soils, and ecosystems in general, IPNI has determined the balances of multiple crop nutrients, including N, in the United States for the agricultural census years from 1987 to 2007 (IPNI 2010a). Employing a nutrient use geographic information system approach (NuGIS) enabled estimation of balances at the national, state, county, and 8-digit hydrologic unit level. The NuGIS results for the Census years 1987, 1992, 1997, 2002, and 2007 showed that the US national N balances were 24, 27, 28, 24, and 28 kg N ha^{-1} (21, 24, 25, 21, and 25 lb N ac^{-1}), respectively, on a total cropland basis, or 36, 38, 37, 33, and 36 kg N ha^{-1} (32, 34, 33, 29, and 32 lb N ac^{-1}) on a per harvested cropland basis (IPNI 2011). The NuGIS-based partial N balances for the Agricultural Census years 1987 to 2007 in the United States are considerably lower than N balances that were similarly estimated for 2002 to 2004 in several other developed countries and regions (Netherlands, 229 kg N ha^{-1} [204 lb N ac^{-1}]; Germany, 113 kg N ha^{-1} [101 lb N ac^{-1}]; European Union, 83 kg N ha^{-1} [74 lb N ac^{-1}]; Organization for Economic Cooperation and Development countries, 74 kg N ha^{-1} [66 lb N ac^{-1}]; France, 54 kg N ha^{-1} [48 lb N ac^{-1}]; and the United Kingdom, 43 kg N ha^{-1} [38 lb N ac^{-1}]) and similar to the N balance for Canada (35 kg N ha^{-1} [31 lb N ac^{-1}]) (OECD 2008).

Roberts et al. (2010) estimated partial N balances in their investigations of N management on 182 response plots in 16 fields in Missouri and found that when the producer-applied N rate and the economic optimum N rate were equivalent, the amount of N required just for soil N maintenance was approximately 40 kg N ha^{-1} (36 lb N ac^{-1}). When the NuGIS partial N balances (IPNI 2011) are compared with these on-farm research results from Missouri, one might conclude that US farmers, in general, may not be applying N rates considerably in excess of crop demand or economic optima, as is often perceived. Results of several studies indicate that partial N balances on tile-drained Mollisols in the Upper Mississippi River Basin supporting corn and soybeans may now actually be negative, when riverine export of N is considered (David et al. 2010). Farmers in the Upper Mississippi River Basin "are using the same amount of N as they were 30 years ago and getting much higher corn yields," according to the lead author of that Mississippi River Basin N study (CTIC 2011). The real challenge in reducing NO_3-N losses from corn–soybean cropping systems in much of the United States is with management of the subsurface drainage systems (David et al. 2010), and the cropping system N loss problems may not be remedied simply by reducing fertilizer N rates. This assertion is supported by the results of a long-term agricultural practice evaluation in the Raccoon River Watershed in Iowa, which indicated there was no significant relationship between river N loads and the applied N rates over a 30-year period (Hatfield et al. 2009).

SUMMARY AND CONCLUSION

A great deal of work remains to identify and implement agricultural nutrient management practices that can improve crop yields and quality in an economic manner, while also reducing the impact of N on the environment. There are significant hurdles which must be overcome to help farmers, and others within the agricultural community and society at large understand and value improved in-field and on-farm, site-specific nutrient management to help minimize the direct and indirect emissions of N_2O. All principles of 4R Nutrient Stewardship—not just N rate management—need to be employed on more lands by US farmers to achieve better N use efficiency and effectiveness. The skills and management advice of certified crop advisers, extension workers, and other agricultural professionals should be increasingly sought by more farmers as they strive to achieve economic, social, and environmental objectives. To sum up, we quote Cassman et al. (2003):

> ...an environmentally proactive agriculture will be required to meet food demand and protect natural resources and environmental quality. It will require policies and markets that direct intensification to existing prime agricultural land while avoiding expansion of cultivated area into natural ecosystems. It also will require substantial investments in research and extension to support scientific advances and timely development and adoption of innovative new technologies.

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