The Need for a Coupled Human and Natural Systems Understanding of Agricultural Nitrogen Loss

DIANA STUART, BRUNO BASSO, SANDY MARQUART-PYATT, ADAM REIMER, G. PHILIP ROBERTSON, AND JINHUA ZHAO

Reactive nitrogen loss from agricultural fertilizer use remains a crucial environmental problem in the United States, contributing to ecosystem degradation and global climate change. This intractable problem requires a coupled human and natural systems approach that combines biophysical, sociological, and economic knowledge into an integrative analysis. Much is known about the biogeochemistry of nitrogen and agricultural nitrogen loss; however, much is not known about how soil variability and climate change will affect farmer decisionmaking. Although it is widely understood that personal values and beliefs, social norms, economics, and policies influence farmer decisionmaking, very little is known about decisionmaking specific to fertilizer management. In addition, little is known about the socioeconomic influences on decisionmaking across scales and how ecological change is perceived and responded to. Combining sociological, economic, and biophysical knowledge can provide key insights regarding how these factors interact and can support more effective strategies to address this persistent problem.

Keywords: agriculture production, agroecosystems, interdisciplinary science, natural resources, complex systems

Largely because of its extensive use in agriculture, excess reactive nitrogen (N) remains one of the most significant environmental problems today. N is an essential constituent of amino acids, required for the assembly of proteins and for the nucleotides that store and process all genetic information. Humans depend on agricultural systems to provide most of their daily protein needs, prompting Liebig (1840) to quip that agriculture's principal objective is the production of digestible N. Today's intensive agriculture is built on a foundation of N augmentation via the use of synthetic fertilizers and the cultivation of nitrogen-fixing crops on a massive scale. The global anthropogenic inputs of N, 195 teragrams per year, now exceed background, preindustrial inputs from terrestrial biological N fixation by a factor of three (Vitousek et al. 2013). Most of this N is for agricultural purposes. Excess N in the environment contributes to aquatic and marine eutrophication and hypoxia, atmospheric smog, and radiative forcing in the atmosphere. The US Environmental Protection Agency has cited nitrate leaching from agriculture as one of the nation's most significant water pollution problems. Agriculture also releases approximately 70% (USEPA 2009) of US emissions of nitrous oxide (N_2O), an important greenhouse gas that also contributes to stratospheric ozone depletion.

Excess N in the environment is a result of a complex set of human behaviors interacting with natural processes and therefore represents a compelling example of a coupled human and natural systems (CHANS) problem. Attention to CHANS research has increased in recent years as scientists increasingly recognize the interconnectedness and interdependence of social and ecological systems (Liu et al. 2007, Collins et al. 2011). CHANS are complex and adaptive and require interdisciplinary perspectives and research approaches (Roy et al. 2013). Recent examples include work on ecosystem services (Liu et al. 2007, Alberti et al. 2011), land use and land-cover change (Brown et al. 2008, Turner et al. 2007), and cities (Ruddell et al. 2010, Grimm et al. 2013). Taken together, these examples demonstrate that the complexity of the issues investigated—from problem conceptualization to data integration and analysis—cannot be understood without linking human and natural systems (Liu et al. 2007). CHANS and other forms of interdisciplinary research also face challenges, including institutional barriers to cross-disciplinary research (such as lack of incentives in tenure and promotion and narrowly-defined research funding opportunities), differences in terminology (Roy et al. 2013), and epistemological differences among disciplines (Miller et al. 2008).

BioScience 65: 571–578.
doi:10.1093/biosci/biv049

Here, we explore excess N in the environment using a CHANS approach. Understanding the excess reactive N problem requires knowledge about the biophysical mechanisms by which N flows through the environment as well as the socioeconomic factors that shape farmer decisionmaking regarding N management. Biophysical scientists have investigated N in the environment for more than a century, and although there are still uncertainties, the basic biophysical factors that affect N pools and flows are mostly well understood. More questions remain regarding the human dimensions of excess N. As N use in global agriculture continues to increase, we believe it useful—and, indeed, necessary—to gain a systems-based understanding of the social, economic, and biophysical factors that contribute to this persistent and pernicious environmental problem. Here, we examine what is known about this system from a CHANS perspective and highlight areas for future study.

Fertilizer use and excess reactive nitrogen

Global N fertilizer use has increased approximately tenfold between 1950 and 2008 as N fertilizers have become a pervasive feature of modern crop management (Robertson and Vitousek 2009). The net benefit to humans of this additional N is immense: It has enabled substantially more food to be grown on a given area of land, thereby increasing human carrying capacity, enabling unprecedented increases in human population and welfare, and sparing lands that would otherwise have been cleared for cultivation–contributing indirectly to biodiversity conservation and climate mitigation (Burney et al. 2010). However, the environmental consequences of this additional N are also substantial. Increased application and persistent inefficiencies in N use have resulted in significant environmental and social impacts (Mosier et al. 2001).

While reactive N losses from cropping systems is one of the most widely recognized environmental problems today (Davidson et al. 2012), it is also one of the most difficult to abate. Less than 40% of the N fertilizer added to most annual grain crops is taken up by the crop (Cassman et al. 2002). The remainder is available for loss through air, surface water, or groundwater pathways (Follett and Delgado 2002). Only where soil organic matter is accumulating, such as in permanent no-till or cover-cropped systems (e.g., Syswerda et al. 2011), is a small portion retained organically bound.

N lost as nitrate pollutes groundwater and surface waters, contributing to human health risks (Peel et al. 2013) and freshwater eutrophication (Conley et al. 2009). Largely because of fertilizer use, approximately 60% of coastal rivers and bays in the United States have been degraded by nutrient pollution (Howarth et al. 2002). Eventually, much of the agricultural nitrate reaches the coastal ocean, where it can cause hypoxic or dead zones that harm coastal fisheries (Goolsby et al. 1999). Nitrogen lost as ammonia enters the atmosphere to be deposited downwind as unwanted wet or dry deposition (Simpson et al. 2011). Nitrogen lost as N oxides to the atmosphere contributes to smog formation and becomes acid rain (Jaeglé et al. 2005). Finally, N lost as the greenhouse gas (GHG) nitrous oxide (N_2O) contributes to climate change (Pachauri and Resinger 2007). N_2O is a powerful warming agent: Over a 100-year period, a kilogram (kg) of N_2O is approximately 300 times more effective at heating the atmosphere than a kg of carbon dioxide.

Why is so much N lost? The primary reason is low N-use efficiency (NUE) at both the plant and cropping system scales. At the plant scale, NUE refers to the amount of N used by a plant to accumulate biomass, often measured as the carbon to N ratio of plant tissue. At the field scale, when soil organic and inorganic N pools are in steady state, NUE refers to the amount of yield per unit of added N, or the proportion of applied N that is removed in harvested biomass (Robertson and Vitousek 2009). In other words, more N-efficient cropping systems produce more biomass with less fertilizer N.

In the United States, about 50% of fertilizer N is applied to corn, 11% to wheat, 10% to turf, and 3% to cotton, with the remainder shared by a number of small grain and horticultural crops (ERS 2012). Corn illustrates the NUE conundrum. Corn puts on most of its biomass during a 6-week period of exponential growth, during which N uptake demands can reach an astonishing 4 kg N per hectare (ha) per day (Robertson 1997). This rate is sustained for 3–4 weeks, after which N uptake falls to nil over the following 2–3 weeks as N in vegetated tissues is remobilized to grain. This demand, though of a relatively short duration, cannot be met solely by the microbial mineralization of soil organic matter, which might provide about 1 kg N per ha per day under favorable conditions. Moreover, because at the plant scale, corn has an intrinsically low NUE (Below et al. 2007), about twice as much N must be available in the soil solution as the plant is capable of extracting. Consequently, sufficient N must be applied to soil before the period of high growth to ensure that enough is available when the corn needs it. Therefore, the soil solution is usually awash in N from the point when the fertilizer is applied until it is taken up by the crop or lost to various environmental fates.

Strategies are available to minimize N loss from cropped fields. Agronomists promote a *4R* strategy for managing N (IFA 2009): applying N at the right rate, at the right time, in the right place, and in the right formulation. Applying no more N than the crop needs (right rate) is the most obvious strategy for conserving N. Applying it closer to when the crop needs it (right time) is another, as is applying it close to growing plants (e.g., by banding fertilizer within the row close to roots) and on the basis of soil fertility differences (e.g., by applying N at variable rates across individual fields; right place). Likewise, applying N in a chemical form that is not easily lost conserves more for plant use (right formulation). A variety of technologies exist for conserving fertilizer N (Robertson and Vitousek 2009), including N-rate calculators, variable-rate applicators that apply N differentially across a field on the basis of past yield patterns or on-the-go leaf spectroscopy, and polymer coatings that dissolve to release N to the soil solution as soil conditions become

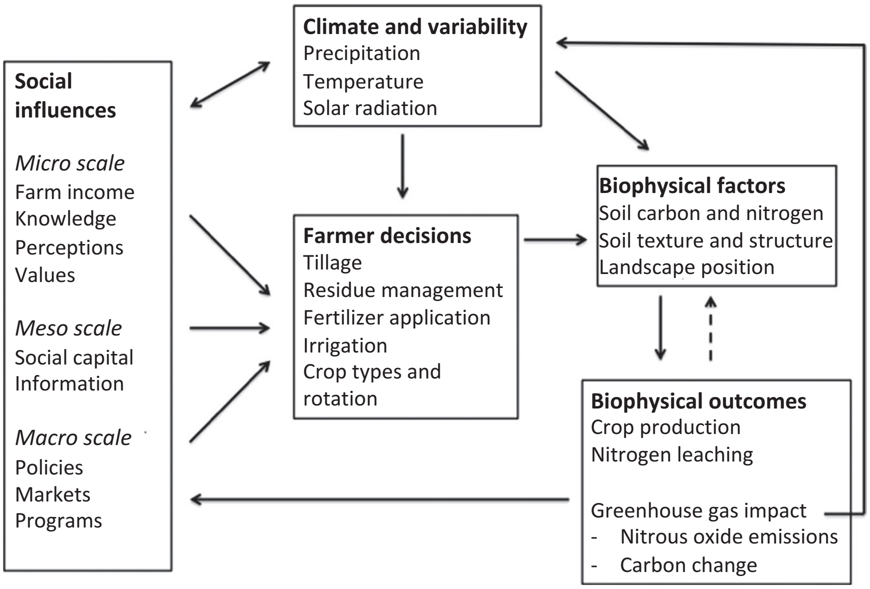

Figure 1. The coupled natural and human nitrogen cycle in agriculture.

favorable for plant growth. However, the adoption rate for these technologies is uniformly low. Most US farmers are not applying these strategies, and crops continue to be fertilized in excess (Ribaudo et al. 2011). For example, few farmers use soil tests prior to fertilizer application and approximately 1/3 of Midwest farmers still apply N fertilizer to corn the prior fall, 6–7 months before the crop needs it (Ribaudo et al. 2011). Although considerable effort continues to be invested in developing strategies to apply N fertilizer more precisely, relatively little is known about how farmers make decisions about N fertilizer application.

Understanding the use and misuse of N fertilizer and its ecological consequences presents a classic CHANS problem (figure 1). Water pollution and GHG emissions are linked to individual human decisions, and these decisions are likely increasingly affected by the associated ecological changes. Because this system has serious ecological and social impacts, it is crucial to explore the human–environment interactions that prevent a tighter coupling of the biogeochemical N cycle. Certainly, it is not a biogeochemical issue alone: We know the major pathways of N gain and loss from cropping systems as well as the biogeochemical controls on most internal transformations. Nor is it strictly a social issue: We know that farmers make decisions that balance competing needs for income, social acceptance, and environmental stewardship. We also know that farmer decisions regarding fertilizer are embedded within a larger context and are not only tied to farm-level factors but are also affected by regional, national, and international market conditions and agricultural policies.

At its core, then, excess N is an interface issue: Understanding the human–natural system interface is crucial to identifying workable solutions to the reactive N problem. This approach requires assessing what we know about the biophysical and socioeconomic factors in this system and then integrating these pieces into a comprehensive understanding of their links. Below, we discuss the current state of understanding regarding the natural and social aspects of this CHANS application and conclude by highlighting areas for future research and the integration of biophysical and socioeconomic findings. Only by combining biogeochemical, economic, and social factors will we sufficiently understand how changes in the future might reshape relationships within this coupled system in new ways.

The biogeochemical challenge

Arguably, enough is known about the biophysical basis of N loss from field crop systems that the only real impediment to enacting solutions is overcoming deployment barriers that are largely economic and social (Robertson and Vitousek 2009). This is true, however, only up to a point: Although we understand much about fundamental N-cycle processes and how they interact with other parts of the biophysical environment, including soil, plant, and climate variables, we nevertheless do not fully understand how key portions will respond to climate change (Robertson et al. 2012) nor how field-scale spatial and temporal variability will affect this response. Uncertainty associated with these responses makes formulating an integrated biophysical, social, and economic understanding substantially difficult.

Of particular import is the uncertainty associated with the temporal and spatial variability of N-cycle responses and, in particular, those responses that interact to affect field-scale NUE (e.g., Basso et al. 2013). This is evident in the response of crop yield to fertilizer-N additions across fields and years. Hundreds of N-response experiments across the upper Midwest over the past few decades have established agronomically optimum N needs for corn across representative soil types and climate years. Results now inform the basis for Maximum Return to Nitrogen (MRTN; Sawyer et al. 2006) university fertilizer recommendations across seven states (Iowa, Illinois, Indiana, Michigan, Minnesota, Ohio, Wisconsin). The MRTN approach to N fertilizer recommendations uses asymptotic yield curves together with fertilizer and expected grain prices to define economically optimum N rates for a particular geographic area.

MRTN recommendations average a substantial amount of annual variability into a single location-specific recommendation. Although in many sites, the response is relatively stable from year to year (annual precipitation differences do not much affect optimum N rates; e.g., Dharmakeerthi et al. 2006), in other sites, year-to-year variability is significant and sometimes substantial (e.g., Bundy et al. 2011, Basso et al. 2012). It may be that part of this stability is soil related: Different soils have the capacity to differentially buffer precipitation variability on crop growth and N supply. For example, a soil profile that enters the growing season fully charged with overwinter water will hold a different amount of water depending on soil structure and soil organic matter (SOM) at different profile depths. Whereas a southwest Michigan Alfisol will typically hold 130 millimeters (mm) of water in the rooting zone, an Iowa Mollisol will hold 250 mm, and more stored water at the start of the growing season provides resilience against drought, lessening its effects on final yields. At the same time, soil water interacts with SOM to buffer crop N needs: In years with growing season rainfall that fully meets crop water needs, the additional rainfall will also stimulate microbes to mineralize more N from SOM, making up what might otherwise be a fertilizer-N deficit—so long as sufficient SOM is available. There has so far been little effort to parse the effects of a soil structure × SOM interaction on N-response stability. Broadly speaking, geomorphologic (but not SOM) differences are embedded in the different MRTN recommendations for different states and regions. However, theory suggests that the interaction of soil structure and SOM could be more important than either factor alone, and knowledge of this relationship could significantly reduce the temporal uncertainty associated with N response rates and regional recommendations for farmers.

An additional overarching source of future uncertainty relates to climate change. Midwest climates are changing: Annual temperatures have increased 0.6 degrees Celsius on average over the past century and at a faster rate in recent decades, mostly as elevated wintertime and nighttime temperatures (Pryor and Barthelmie 2012). The frost-free growing season has so far lengthened by two weeks with earlier springs and later first frosts (Schoof 2009), and plant-hardiness zones have shifted northward (USDA 2012). In addition—and perhaps more importantly for agricultural N loss (Robertson et al. 2012)—climate variability has also increased, expressed as more frequent winter thaws and summer heat waves and more variable precipitation (Villarini et al. 2011), with spring rainfall more frequent and growing season rainfall occurring in larger, more separated events.

A particular vector both affected by and affecting climate is N_2O loss. Agricultural soils are responsible for about 60% of global anthropogenic N_2O production (Pachauri and Resinger 2007), and changing temperatures and precipitation patterns may have particularly important impacts on N_2O fluxes. More wintertime thaws with less snow cover can substantially increase wintertime emissions (figure 2), and a greater frequency of dry soils preceding growing season rain events can significantly increase the mole ratio of N_2O to N_2 (Bergsma et al. 2002), potentially leading to greater growing-season N_2O emissions. Especially in light of new carbon market protocols that reward farmers for lowering their N_2O emissions (Millar et al. 2010), understanding these future impacts will be important for farmer decisionmaking.

The socioeconomic challenge

Farmer fertilizer decisions are embedded within a complex social and economic system that influences not only fertilizer use but also related behaviors, such as technology adoption, crop choices, production decisions, and the use of best management practices. These field- and farm-level decisions are influenced by state and federal farm policies, as well as by global market forces. As with biophysical systems, social and economic systems are subject to feedback. Particularly important in the context of excess N is the feedback between ecological change (e.g., nitrogen leaching and eutrophication) and federal policies.

The problems associated with excess N are well known and have resulted in policy changes. For example, in recent decades, the federal government has significantly increased investment in working-lands conservation programs that address ongoing environmental problems stemming from agriculture, including N leaching (Reimer and Prokopy 2014). The federal government has attempted to address hypoxia in the Gulf of Mexico directly through policies and programs aimed at reducing fertilizer application and altering land use (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force 2008). Other indirect feedback exists as well, including federal policies that affect crop choices (e.g., direct payments) and production decisions (e.g., bioenergy policy).

Because the environmental impacts of agriculture emerge from the cumulative actions of individual farmers, social scientists—including sociologists and economists—have explored factors influencing the adoption of environmentally friendly farming practices in the United States. Although some of this literature has focused on aspects related to farm N management, there is still uncertainty surrounding how farmers make fertilizer decisions, including application methods and rates. Much of the social and behavioral research regarding farmer decisionmaking has been framed around the adoption of conservation practices. A number of studies have explored farmer adoption of comprehensive nutrient management plans, soil testing, or variable-rate fertilizer application (Prokopy et al. 2008). Most have failed to explore N management in its entirety, and because farmers may not view application decisions specifically through an environmental lens, studies focusing primarily on the environmental aspects of decisionmaking may miss important behavioral determinants.

The majority of the sociological adoption literature has focused on microscale (individual farmer or farm) or mesoscale (community level) factors. In this literature, which broadly explores adoption of conservation practices in general, the importance of demographic, farm structure, and socioeconomic variables, as well as the role of personal values and beliefs, access to information, risk perceptions, and other individual characteristics, has been exhaustively examined (e.g., Napier et al. 1986, Napier and Camboni 1988). Others have scaled their analyses beyond individual farmers to explore the spread of information and the importance of social networks and

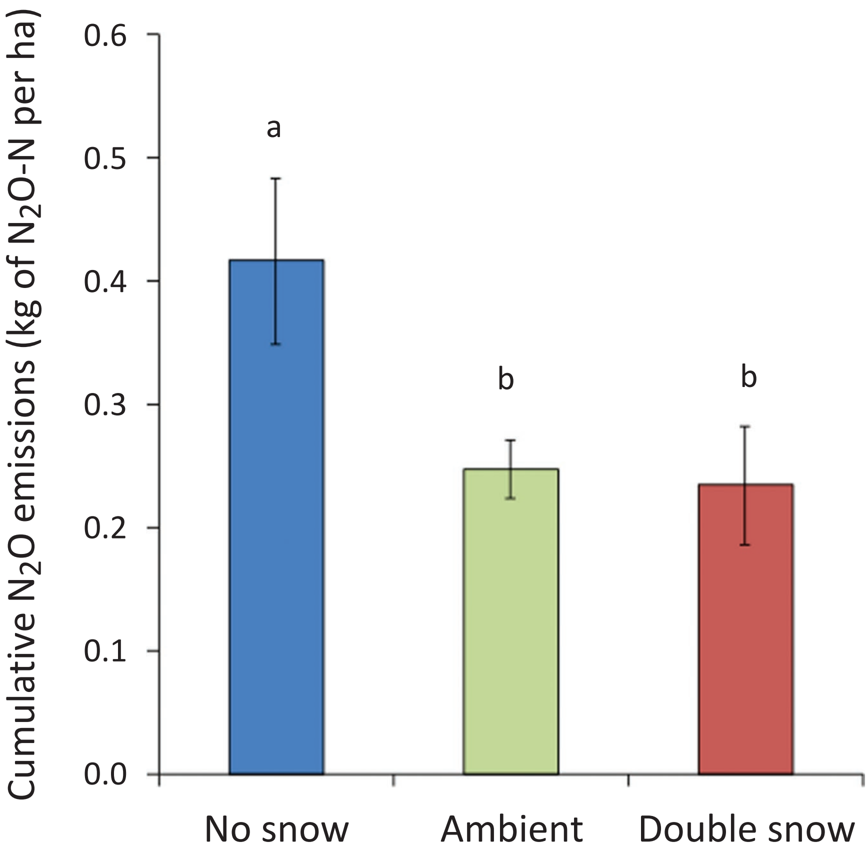

Figure 2. The N_2O response to winter thaws that reduce snow cover in automated chambers (Ruan 2014). Abbreviations: ha, hectares; kg, kilograms; N_2O-N, nitrous oxide nitrogen.

social capital, including trust and reciprocity (Lubell and Fulton 2007).

Specific to N or nutrient management, social science studies have focused on the adoption of soil testing (Bosch et al. 1995, Khanna 2001), the incorporation of variable-rate application or precision-agriculture techniques (Daberkow and McBride 2003), and the reduction of inputs (Lasley et al. 1990, McGuire et al. 2012). These studies have identified a number of important factors influencing individual behaviors, including technical capacity (Khanna 1999); land tenure (owning versus renting; Bosch et al. 1995); and attitudes toward technology, the environment, and personal safety (Lasley et al. 1990). Economics remains an important factor influencing application because N fertilizer represents a major cost (approximately 45%) of corn production. Nevertheless, relative to the price of corn, until very recently the cost has not much increased over the past 20 years, even though in absolute terms, N fertilizer is two to three times more expensive than it was 15 years ago (Robertson and Vitousek 2009).

A preliminary study focusing on corn farmers in southwest Michigan indicates that knowledge and sources of information may represent important factors influencing the application of N fertilizer (Stuart et al. 2012). Although this study included only four Michigan counties, the results highlight several important factors: For example, a mail survey indicated that most corn farmers (70%) in the region received the bulk of their information about N application rates from fertilizer dealers and that a majority ranked fertilizer dealers (37%) and seed company agronomists (18%) together as their most important sources of information (figure 3). The mail survey also revealed that 77% of respondents never used university fertilizer recommendations. Results from the survey, interviews, and focus groups indicate that very few farmers know that N fertilizer is linked to GHG emissions and that only 5–10% of survey respondents had ever used tools such as soil tests and delayed-release products.

Stuart and colleagues (2012) also explored potential policy options aimed at reducing N inputs. Farmers were asked to indicate their interest in potential offsets programs to pay farmers for reducing N application: Approximately 50% of farmers surveyed stated that they might be interested in such a program. In a broader survey about ecosystem services in general, Ma and colleagues (2012) found that 90% of surveyed corn–soybean farmers in Michigan were willing to accept some level of payment for services, including reduced fertilizer application. Additional research over a broader geographical area is needed to further explore these trends and the relative importance of these and other factors influencing farmer decisionmaking.

Much of the economics literature has focused on macroscale factors, including policy and market conditions, and posits that a farmer will adopt a new technology or practice if the benefits of doing so exceed the costs (e.g., Feder et al. 1985). Such threshold models have been applied to a growing body of literature on adoption of conservation technologies (Lichtenberg et al. 2010), such as low- or no-tillage practices (Knowler and Bradshaw 2007) and irrigation technologies (Schoengold and Zilberman 2007). Within this framework, the diffusion of a new technology is driven mostly by farmer heterogeneities (e.g., low-cost farmers adopt first, followed by high-cost farmers) or by network externalities, such as reduced equipment costs over time. Increasingly, the economics literature recognizes the importance of information and learning in the adoption of both farm innovations in general and conservation practices specifically (e.g., Foster and Rosenzweig 1995, Zhao 2007), but empirical studies

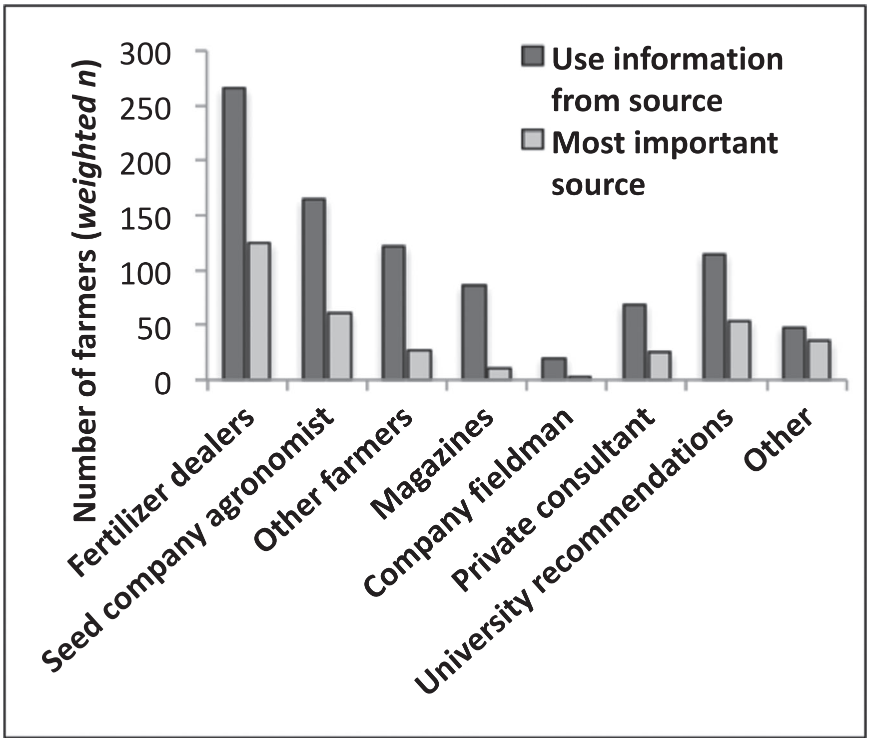

Figure 3. The sources of information farmers in southwest Michigan use to make nitrogen fertilizer application decisions (Stuart et al. 2012).

have been limited. Such studies have also failed to incorporate other micro- and mesoscale factors emphasized in sociological studies. Like the sociological studies, economics research exploring N management in its entirety is lacking. Studies framed around specific N-management practices (e.g., soil testing, variable-rate fertilizer applications) fail to elaborate on how farmers make nutrient-management decisions in general, including how they evaluate different sources of information; how they determine application rates, methods, and timing; and how macroscale social and financial factors influence individual decisionmaking.

Even when ideas about environmental stewardship may resonate with an individual farmer (on the basis of personal beliefs and values), management decisions can be constrained by larger-scale economic or policy factors (Hendrickson and James 2005, Stuart 2009). Farmers who have participated in environmental efforts may abandon such endeavors because of changing market conditions. For example, some US corn farmers significantly changed their practices in response to demands for ethanol (Stuart and Gillon 2013). Studies indicate that global policies and market conditions have an increasingly important role in shaping farmer decisionmaking regarding the environment (Atwell et al. 2009). To fully understand the factors driving farmer decisionmaking, we must understand the interactions of influences across multiple scales, not just the micro-, meso-, and macroscale factors separately.

We know that a variety of factors influence N-fertilizer decisionmaking, including awareness of environmental issues, access to information and tools, sources of information, social capital (including trust and reciprocity), personal values and beliefs, and government policies and programs. Farmers are making decisions about N-fertilizer application on the basis of a complex set of factors influenced by decisions at multiple scales. However, studies have continued to include only a few of these factors at a time. CHANS research that focuses on how combined factors across scales interact to shape the context of N–application rate decisions is needed. In addition, farmer decisionmaking is linked to biophysical factors, both directly through experienced ecological change and indirectly through social and policy changes at the meso- and macroscales, respectively. Most farmers are astute observers, and their perceptions of environmental change will increasingly affect decisionmaking.

New policies and programs that emerge as result of environmental change will also increasingly shape land management decisions. Continued policy emphasis on environmental problems linked with agriculture, including excess N, will influence farmer behavior directly by incentivizing particular behaviors and indirectly via production decisions as well as land use. To understand excessive N as a CHANS, future researchers will need to emphasize such links between these social and biophysical factors and examine these relations across scales.

Conclusions

Excess reactive N in the environment is a result of cumulative individual behaviors, as influenced by factors at different spatial scales, interacting with the biophysical properties of N. As is illustrated in figure 1, many factors are involved in these relationships. Although we know much about the biophysical basis for excess N, too little is known about forthcoming interactions with climate change and, in particular, about the effects of increasing climate variability. Social science research indicates important factors influencing farmer decisionmaking; however, in general, our understanding of the social, economic, and behavioral aspects of N loss remains insufficient for developing effective management and policy solutions. For instance, we do not know the extent of the use of N-efficiency strategies across the United States, what factors across different scales influence N-management decisions, and how farmers might perceive different policy approaches. Although researchers continue to examine N loss from biophysical and socioeconomic perspectives separately, what is most needed is an understanding of the interdependencies between farmer decisionmaking and biogeochemical cycles.

A CHANS approach provides the opportunity to analyze the social and biophysical factors influencing N loss together through an integrative model. In particular, multilevel structural equation modeling (Preacher et al. 2010) can be used to examine interdependencies, reciprocity, and feedback effects among select biophysical and social components of the system. Integrative models can be used to link biophysical, social, and economic data together to comprehensively analyze the factors influencing decisionmaking and environmental impacts, what factors are most influential, and how interactions may change over time. Identifying the recursive relations and understanding the interdependencies and feedback of these social and biophysical factors are crucial for identifying solutions to the excess N problem. As policymakers move forward with their approaches to address excess N in the environment, CHANS research can provide the insights needed to understand how social and biophysical factors interact across scales. Knowledge of dynamic cross-scale relationships should lead to new, more effective strategies to address this persistent and pervasive problem.

Acknowledgments

This research was supported by the National Science Foundation's (NSF) Dynamics of Coupled Natural and Human Systems program, award number 1313677 with additional support from NSF's Kellogg Biological Station Long Term Ecological Research Site (NSF grant no. DEB 1027253) and Michigan State University AgBioResearch.

References cited

Alberti M, et al. 2011. Research on coupled human and natural systems (CHANS): Approach, challenges, and strategies. Bulletin of the Ecological Society of America 92: 218–228.

Atwell RC, Schulte LA, Westphal LM. 2009. Linking resilience theory and diffusion of innovations theory to understand the potential for perennials in the US corn belt. Ecology and Society 14: 30–47.

Basso B, Sartori L, Cammarano D, Grace P, Sorensen C, Fountas S. 2012. Environmental and economic evaluation of N fertilizer rates in a maize crop in Italy: A spatial and temporal analysis using crop models. Biosystems Engineering 113: 103–111.

Basso B, Cammarano D, Fiorentino D, Ritchie JT. 2013. Wheat yield response to spatially variable nitrogen fertilizer in Mediterranean environment. European Journal of Agronomy 51: 65–70.

Below FE, Uribelarrea M, Ruffo M, Moose SP, Becker AW 2007. Triple-Stacks, Genetics, and Biotechnology in Improving Nitrogen Use of Corn. Pages 5–13 in Schwab G, ed. Proceedings of the North Central Extension-Industry Soil Fertility Conference, vol. 23. Department of Crop Science, University of Illinois at Urbana–Champaign.

Bergsma TT, Robertson GP, Ostrom NE. 2002. Influence of soil moisture and land use history on denitrification end products. Journal of Environmental Quality 31: 711–717.

Bosch DJ, Cook ZL, Fuglie KO. 1995. Voluntary versus mandatory agricultural policies to protect water quality: Adoption of nitrogen testing in Nebraska. Review of Agricultural Economics 17: 13–24.

Brown DG, Robinson DT, An L, Nassauer JI, Zellner M, Rand W, Riolo R, Page SE, Low B, Wang Z. 2008. Exurbia from the bottom up: Confronting empirical challenges to characterizing a complex system. Geoforum 39: 805–18.

Bundy LG, Andraski TW, Ruark MD, Peterson AE. 2011. Long-term continuous corn and nitrogen fertilizer effects on productivity and soil properties. Agronomy Journal 103: 1346–1351.

Burney JA, Davis SJ, Lobell DB. 2010. Greenhouse gas mitigation by agricultural intensification. Proceedings of the National Academy of Sciences 107: 12052–12057.

Cassman KG, Dobermann A, Walters DT. 2002. Agroecosystems, nitrogen-use efficiency, and nitrogen management. Ambio 31: 132–140.

Collins SL, et al. 2011. An integrated conceptual framework for long-term social–ecological research. Frontiers in Ecology and the Environment 9: 351–357.

Conley DJ, Paerl HW, RW H, Boesch DF, Seitzinger SP, Havens KE, Lancelot C, Likens GE. 2009. Controlling eutrophication: Nitrogen and phosphorus. Science 323: 1014–1015.

Daberkow SG, McBride WD. 2003. Farm and operator characteristics affecting the awareness and adoption of precision agriculture technologies in the US. Precision Agriculture 4: 163–177.

Davidson EA, et al. 2012. Excess Nitrogen in the US Environment: Trends, Risks, and Solutions. Ecological Society of America. Issues in Ecology Report no. 15.

Dharmakeerthi RS, Kay BD, Beauchamp EG. 2006. Spatial variability of in-season nitrogen uptake by corn across a variable landscape as affected by management. Agronomy Journal 98: 255–264.

[ERS] Economic Research Service. 2012. Fertilizer Use and Price. US Department of Agriculture. (18 October 2012; *www.ers.usda.gov/data-products/fertilizer-use-and-price.aspx*)

Feder G, Just RE, Zilberman D. 1985. Adoption of agricultural innovations in developing countries: A survey. Economic Development and Cultural Change 33: 255–295.

Follett RF, Delgado JA. 2002. Nitrogen fate and transport in agricultural systems. Journal of Soil and Water Conservation 57: 402–408.

Foster A, Rosenzweig M. 1995. Learning by doing and learning from others: Human capital and technological change in agriculture. Journal of Political Economy 103: 1176–1209.

Grimm NB, Redman CL, Boone CG, Childers DL, Harlan SL, Turner, BL II. 2013. Viewing the urban socioecological system through a sustainability lens: Lessons and prospects from the Central Arizona–Phoenix LTER Programme. Pages 217–246 in Singh SJ, Haberl H, Chertow M, Mirtl M, and Schmid M, eds. Long Term Socio-Ecological Research: Studies in Society–Nature Interactions Across Spatial and Temporal Scales. Springer.

Goolsby DA, Battaglin WA, Lawrence GB, Artz RS, Aulenbach BT, Hooper RP, Keeney DR, Stensland GJ. 1999. Fluxes and Sources of Nutrients in the Mississippi–Atchafalaya River Basin: Topic 3 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico. National Oceanic and Atmospheric Administration Coastal Ocean Program. Decision Analysis Series no. 17.

Hendrickson M, James HS. 2005. The ethics of constrained choice: How the industrialization of agriculture impacts farming and farmer behavior. Journal of Agricultural and Environmental Ethics 18: 269–291.

Howarth RW, Sharpley A, Walker D. 2002. Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. Estuaries 25: 656–676.

[IFA] International Fertilizer Industry Association Task Force on Fertilizer Best Management Practices. 2009. The Global "4R" Nutrient Stewardship Framework for Developing and Delivering Fertilizer Best Management Practices. IFA.

Jaeglé L, Steinberger L, Martin RV, Chance K. 2005. Global partitioning of NOx sources using satellite observations: Relative roles of fossil fuel combustion, biomass burning, and soil emissions. Faraday Discussions 130: 407–423.

Khanna M. 1999. Sequential adoption of site-specific technologies and its implications for nitrogen productivity: A double selectivity model. Paper presented at the Annual Meeting of the American Agricultural Economics Association. 8–11 August 1999. Nashville, Tennessee.

Knowler D, Bradshaw B. 2007. Farmers' adoption of conservation agriculture: A review and synthesis of recent research. Food Policy 32: 25–48.

Lasley P, Duffy M, Kettner K, Chase C. 1990. Factors affecting farmers' use of practices to reduce commercial fertilizers and pesticides. Journal of Soil and Water Conservation 45: 132–136.

Lichtenberg E, Shortle J, Wilen J, Zilberman D. 2010. Natural resource economics and conservation: Contributions of agricultural economics and agricultural economists. American Journal of Agricultural Economics 92: 469–486.

Liebig J. 1840. Chemistry in Its Application to Agriculture and Physiology. Taylor and Walton.

Liu J, et al. 2007. Coupled human and natural systems. Ambio 36: 639–649.

Lubell M, and Fulton A. 2007. Local diffusion networks act as pathways to sustainable agriculture in the Sacramento River Valley. California Agriculture 61: 131–137.

Lubowski RN, Plantinga AJ, Stavins RN. 2008. What drives land-use change in the United States? A national analysis of landowner decisions. Land Economics 84: 529–550.

Ma S, Swinton SM, Lupi F, Jolejole-Foreman CB. 2012. Farmers' willingness to participate in payment-for-environmental-services programmes. Journal of Agricultural Economics 63: 604–626.

Madrid C, Cabello V, Giampietro M. 2013. Water-use sustainability in socioecological systems: A multiscale integrated approach. BioScience 63: 14–24.

McGuire J, Morton LW, Cast AD. 2013. Reconstructing the good farmer identity: Shifts in farmer identities and farm management practices to improve water quality. Agriculture and Human Values 30: 57–69.

Millar N, Robertson GP, Grace PR, Gehl RJ, Hoben JP. 2010. Nitrogen fertilizer management for nitrous oxide (N_2O) mitigation in intensive corn (maize) production: An emissions reduction protocol for US Midwest agriculture. Mitigation and Adaptation Strategies for Global Change 15: 185–204.

Miller TR, Baird TD, Littlefield CM, Kofinas G, Chapin FS II, Redman CL. 2008. Epistemological pluralism: Reorganizing interdisciplinary research. Ecology and Society 13 (art. 46).

Mississippi River/Gulf of Mexico Watershed Nutrient Task Force. 2008. Gulf Hypoxia Action Plan 2008 for Reducing, Mitigating, and Controlling Hypoxia in the Northern Gulf of Mexico and Improving Water Quality in the Mississippi River Basin. Mississippi River/Gulf of Mexico Watershed Nutrient Task Force. (2 April 2015; *http://water.epa.gov/type/watersheds/named/msbasin/upload/2008_8_28_msbasin_ghap2008_update082608.pdf*)

Mosier AR, et al. 2001. Policy implications of human-accelerated nitrogen cycling. Biogeochemistry 52: 281–320.

Napier TL, Camboni SM. 1988. Attitudes toward a proposed soil conservation program. Journal of Soil and Water Conservation 43: 186–191.

Napier TL, Camboni SM, Thraen CS. 1986. Environmental concern and the adoption of farm technologies. Journal of Soil and Water Conservation 41: 109–113.

Pachauri RK, Resinger A, eds. 2007. Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II, and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change Intergovernmental Panel on Climate Change.

Peel JL, Haeuber R, Garcia V, Russell AG, Neas L. 2013. Impact of nitrogen and climate change interactions on ambient air pollution and human health. Biogeochemistry 114: 121–134.

Preacher K, Zyphur, M, Zhang, Z. 2010. A general multilevel SEM framework for assessing multilevel mediation. Psychological Methods 15: 209–233.

Prokopy LS, Floress K, Weinkauf DK, Baumgart-Getz A. 2008. Determinants of agricultural BMP adoption: Evidence from the literature. Journal of Soil and Water Conservation 63: 300–311.

Pryor SC, Barthelmie RJ. 2012. The midwestern United States: Socioeconomic context and physical climate. Pages 12–47 in Pryor SC, ed. Climate Change in the Midwest: Impacts, Risks, Vulnerability, and Adaptation. Indiana University Press.

Reimer A, Prokopy L. 2014. One federal policy, four different policy contexts: An examination of agri-environmental policy implementation in the midwestern United States. Land Use Policy 38: 605–614.

Ribaudo M, Delgado J, Hansen L, Livingston M, Mosheim R, Williamson J. 2011. Nitrogen in Agricultural Systems: Implications for Conservation Policy. US Department of Agriculture. Economic Research Report no. ERR-127.

Robertson GP. 1997. Nitrogen use efficiency in row crop agriculture: Crop nitrogen use and soil nitrogen loss. Pages 347–365 in Jackson LE, ed. Ecology in Agriculture. Academic Press.

Robertson GP, Vitousek PM. 2009. Nitrogen in agriculture: Balancing the cost of an essential resource. Annual Review of Environment and Resources 34: 97–125.

Robertson GP, Klingensmith KM, Klug MJ, Paul EA, Crum JR, Ellis BG. 1997. Soil resources, microbial activity, and primary production across an agricultural ecosystem. Ecological Applications 7: 158–170.

Robertson GP, Bruulsema TW, Gehl R, Kanter D, Mauzerall DL, Rotz A, Williams C. 2013. Nitrogen–climate interactions in US agriculture. Biogeochemistry 114: 41–70.

Roy ED, Morzillo AT, Seijo F, Reddy SMW, Rhemtulla JM, Milder JC, Kuemmerle T, Martin SL. 2013. The elusive pursuit of interdisciplinarity at the human–environment interface. BioScience 63: 745–753.

Ruan L. 2014. Impacts of biofuel crops on greenhouse gas emissions from agricultural ecosystems. Dissertation. Michigan State University.

Ruddell DM, Harlan SL, Grossman-Clarke S, Buyantuyev A. 2010. Risk and exposure to extreme heat in microclimates of Phoenix, AZ. Pages 179–202 in Showalter PS and Lu Y, eds. Geospatial Techniques in Urban Hazard and Disaster Analysis. Springer.

Sawyer JE, Nafziger ED, Randall GW, Bundy LG, Rehm GW, Joern BC. 2006. Concepts and Rationale for Regional Nitrogen Rate Guidelines for Corn. Iowa State University Extension. Publication no. PM2015.

Schoengold K, Zilberman D. 2007. The economics of water, irrigation, and development. Pages 2940–2966 in Evenson R, Pingali P, eds. Handbook of Agricultural Economics: Agricultural Development: Farmers, Farm Production, and Farm Markets. Handbooks in Economics, vol. 3. Elsevier.

Simpson D, et al. 2011. Atmospheric transport and deposition of reactive nitrogen in Europe. Pages 298–316 in Sutton MA, Howard CM, Erisman JW, Billen G, Bleeker A, Grennfelt P, van Grinsven H, Grizzetti B, eds. The European Nitrogen Assessment. Cambridge University Press.

Stuart D. 2009. Constrained choice and ethical dilemmas in land management: Environmental quality and food safety in California agriculture. Journal of Agricultural and Environmental Ethics 22: 53–71.

Stuart D, Gillon S. 2013. Scaling up to address new challenges to conservation on US farmland. Land Use Policy 31: 223–236.

Stuart D, Schewe RL, McDermott M. 2012. Responding to climate change: Barriers to reflexive modernization in US agriculture. Organization and Environment 25: 308–327.

Turner BL II, Lamblin EF, Reenberg A. 2007. The emergence of land change science for global environmental change and sustainability. Proceedings of the National Academy of Sciences 104: 20666–20671.

[USEPA] US Environmental Protection Agency. 2009. Inventory of US Greenhouse Gas Emissions and Sinks: 1990–2007. USEPA. EPA Report no. 430-R-09-004. (2 April 2015; *www.epa.gov/climatechange/Downloads/ghgemissions/GHG2007entire_report-508.pdf*)

Villarini G, Smith JA, Baeck ML, Vitolo R, Stephenson DB, Krajewski WF. 2011. On the frequency of heavy rainfall for the Midwest of the United States. Journal of Hydrology 400: 103–120.

Vitousek PM, Menge DNL, Cleveland CC, Reed SC. 2013. Biological nitrogen fixation: Rates, patterns and ecological controls in terrestrial ecosystems. Philosophical Transactions of the Royal Society B 368 (art. 20130119). (2 April 2015; *http://dx.doi.org/10.1098/rstb.2013.0119*)

Zhao J. The role of information in technology adoption under poverty. Pages 191–203 in Nissanke M, Thorbecke E, eds. The Impact of Globalization on the World's Poor. Palgrave Macmillan.

Diana Stuart (dstuart@msu.edu), Adam Reimer (apreimer@msu.edu), and G. Philip Robertson (robert30@msu.edu) are affiliated with the Kellogg Biological Station at Michigan State University, in Hickory Corners. Bruno Basso (basso@msu.edu) is affiliated with the Department of Geological Sciences, Sandy Marquart-Pyatt (marqua41@msu.edu) with the Department of Sociology, and Jinhua Zhao (jzhao@msu.edu) with the Department of Economics at Michigan State University, in East Lansing.