AGROECOSYSTEM SUSTAINABILITY IN THE MISSISSIPPI RIVER BASIN: ASSESSING ECOLOGICAL AND SOCIAL DRIVERS OF NITROGEN POLLUTION FROM GRAIN FARMS

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AGROECOSYSTEM SUSTAINABILITY IN THE MISSISSIPPI RIVER BASIN: ASSESSING ECOLOGICAL AND SOCIAL DRIVERS OF NITROGEN POLLUTION FROM GRAIN FARMS

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Nitrogen (N) leaching to surface waters from grain farms in the Mississippi River Basin (MRB) is the primary cause of hypoxia in the Gulf of Mexico. I constructed N mass balances for a gradient of farm types to explore the relationship between agroecosystem management practices and potential N loss on 95 farms in the upper MRB. Nitrogen balances ranged from high average annual surpluses (149 kg N $ha^{-1}yr^{-1}$) to large deficits (80 kg N $ha^{-1}yr^{-1}$). Fields with greater than 50% of total N additions from legume N sources, and fields with crop rotations that included both annual and perennial species, were approximately in balance, compared to fertilizer-based practices in corn-soybean rotations with surpluses of 35 kg N $ha^{-1}yr^{-1}$.

To explore how a subset of farmers in Iowa transitioned to practices with the greatest promise for reducing N losses, I analyzed qualitative interviews conducted with farmers between 2008 and 2010. I identified resources and strategies they harnessed to develop opportunities for, and overcome barriers to, transitioning to alternative practices within the context of the industrialization of agriculture in the MRB. To enhance resilience and mitigate risk, alternative farmers increased farm-level biodiversity and enterprise diversity. They developed new competencies such as ecological thinking and cultivated external network linkages with peers, knowledge organizations, and federal policy.

Finally, I conducted a ^{15}N-tracer experiment in Illinois to investigate the biogeochemical mechanisms of one ecological practice that has promise for reducing N losses, use of winter annual cover crops. I applied ^{15}N-labeled ammonium sulfate fertilizer to corn, and compared winter rye (*Secale cereal*) cover and bare fallow treatments following corn harvest. After one

year, total recovery of ^{15}N ranged from 37-45%. Due to unfavorable weather conditions rye biomass was low and little ^{15}N was recovered in the rye. However, the cover crop significantly reduced soil inorganic N pools in the spring (11.1 kg N ha^{-1} in bare fallow compared to 1.9 kg N ha^{-1} with rye cover), by an amount similar in magnitude to total rye N uptake (23.7 kg N ha^{-1}), indicating that cover crops scavenge inorganic N mineralized from soil organic matter pools.

BIOGRAPHICAL SKETCH

Jennifer Blesh Gardner was born and raised in the Northwest corner of Connecticut. She received a B.S. degree in Ecology from the Institute of Ecology at the University of Georgia in Athens, GA, in 2003. Before finishing her undergraduate program, she completed a year of domestic service with AmeriCorps, building homes in partnership with Habitat for Humanity International in Americus, GA. Jennifer traveled around the world on a ship with Semester at Sea during the fall semester of 2001, and after graduate school she plans on returning to Brazil to pursue her interdisciplinary research interests on agrifood systems, environmental sustainability and social equity. After receiving her bachelor’s degree, Jennifer worked for a year and a half as general manager of a café and bakery in Athens, GA, and she hiked the entire Appalachian Trail in summer and early fall of 2004. She moved to Ithaca, NY, in 2005 to pursue graduate studies at Cornell University in agroecology. She received her M.S. in Soil Science from Cornell in May 2008.

For my mother, Carol

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As I write these acknowledgements, interdisciplinary work is rapidly gaining traction in academia, slowly eroding the confines of disciplinary constructs. Successfully conducting interdisciplinary research as these changes unfold required making connections and conversing with a vast range of people. I am therefore indebted to an unusually large community of people who contributed to this work—either directly to the research process or by providing social support to help make my graduate school experience professionally fulfilling, and even fun. I could not have completed this work without such a unique and inspiring community, and I am deeply grateful to everyone who has contributed over the years.

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INTRODUCTION

"Without a minimum of hope, we cannot so much as start the struggle."
Paulo Freire (1995)

"The care of the Earth is our most ancient and most worthy, and after all our most pleasing responsibility. To cherish what remains of it and to foster its renewal is our only hope."
Wendell Berry (1977)

In this introduction I offer a broad conceptual outline of the motivation for and philosophical underpinnings of my work on one specific ecological crisis of agriculture. Today we find ourselves in unique environmental and social conditions where multiple global crises are converging—food, climate, water and energy. We are nearing peak oil, a tipping point for atmospheric CO_2 concentrations and the resulting global climate change. Rapid urbanization as people migrate from rural areas to cities has the potential to create a "planet of slums," and we are seeing malnutrition of a "stuffed and starved" sort (Patel, 2007) in which obesity and hunger coexist. Over the past several decades there has been increasing public recognition that resource-extractive industrial food systems narrowly focused on production have externalized many environmental and social costs and contribute to all of these crises. In addition, agriculture is an inherently interdisciplinary field of study because agricultural practice, by definition, is a hybrid of people and nature. Agrifood systems thus make a highly suitable case for exploring the socioecological drivers of unsustainability, and for seeking out and understanding the rapidly growing "spaces of hope" (Zimmerer, 2006) that can inform transformation toward sustainability.

Given the conditions of the contemporary moment the stakes are high. This fact has led to growing calls for new ways of approaching academic research on agrifood systems. For example, Wes Jackson, founder of the Land Institute in Salina, KS, has called for reframing the conversation around the problem *of* agriculture, rather than focusing on problems *in* agriculture. Similarly, in an analysis of the recent food price crisis of 2008, McMichael chose the phrase "crisis of the food system" over "world food crisis," which better reflected that the crisis was both systemic and historical. How we choose to name and frame these problems matters for the solutions that will be proposed. And due to the inherent complexity of the global food system, solutions to today's crises will require systemic, interdisciplinary perspectives. Therefore, while there will always be a role for curiosity-driven research, more emphasis should be placed on action-oriented research that is engaged with communities—work that explicitly contributes to environmental sustainability and social justice. Doing this requires rejecting false binaries typical of Western thought such as nature/society, or "basic" and "applied" research. Though it is not acknowledged by positivist scientific philosophy, social forces are important in shaping and driving scientists' work and the direction of science (Kuhn, 1962; Latour, 1993). Other agroecologists have pioneered this type of interdisciplinary and systemic approach (e.g., Vandermeer and Perfecto, 1995; Perfecto et al. 2009), though it remains rare for natural scientists to link biophysical and critical social science theories and methods. In this work I aimed to advance an *engaged* agroecology that seeks to understand the sociopolitical relations of food systems and to locate power, and potential leverage points for change, in those relations (Rocheleau, 2008). To do that, in this dissertation I approached one specific agricultural problem—surface water pollution caused by nitrogen leaching losses from grain farms in the upper Mississippi River Basin—as both socioecological and systemic.

Nitrogen is an important nutrient limiting crop productivity that is widely applied to agricultural landscapes to increase yields. Humans have profoundly altered the earth's biogeochemical nitrogen cycle (Galloway and Cowling, 2002) with industrial food production, primarily through excess applications of commercial nitrogen fertilizers and increased cultivation of legume crops such as soybeans (Vitousek et al., 1997). Soluble forms of nitrogen are highly mobile, and a proportion of applied nitrogen leaches to surface waters and is transported to aquatic ecosystems causing nutrient enrichment and oxygen concentrations so low that most marine life cannot survive (popularly called 'dead zones'). These zones are widely distributed in coastal marine environments worldwide. In addition to water pollution, the leakiness of the nitrogen cycle caused by human nitrogen forcing has increased gaseous emissions of nitrous oxide, which contribute to global climate change.

Taking a systemic and socioecological perspective on the problem of nitrogen pollution is an ambitious goal for any one individual. It became a realistic goal for this dissertation, however, because my work was nested within an interdisciplinary research collaboration in the National Science Foundation's Coupled Human and Natural Systems program. The larger research team was composed of academics from disciplines as widespread as political ecology and biogeochemistry, and included agronomists, economists, and a policy organization in Washington, D.C. In this team, we took the problem of nitrogen pollution from grain farms in the upper Mississippi River Basin, and the resulting hypoxic (or, "low oxygen") zone in the Gulf of Mexico, as a case study to investigate socioecological processes at multiple scales in this region that is dominated by agricultural landscapes. Rather than treating human management as an external force, or treating *social* and *ecological* as separate domains of reality, this project explicitly integrated human and natural systems, recognizing that they can co-evolve towards or away from sustainability (Norgaard, 1994).The integrated research aimed to test hypotheses about mechanisms that couple social and ecological systems in intensive grain cropping systems in the MRB, and to inform development of policy instruments to reward farmers for services that improve environmental quality.

Within this context, my dissertation work linked social and ecological knowledge in multiple ways. I conducted two studies grounded in ecological and biogeochemical theory—an observational study using mass balance methods, and a field experiment using stable isotope methods—to improve basic understanding of the flows and fates of nitrogen at a field scale. The mass balance study spanned a large number of working grain farms in the upper Mississippi River Basin. The outcomes of that work therefore directly reflect the ways in which farmers currently manage their land given the existing sociopolitical and economic context of the region, and are highly relevant for development of conservation policies targeting the most problematic areas of the basin. I used the outcomes of that work to inform the design of a social science study seeking to understand how a small subset of farmers in the Corn Belt of the U.S. had successfully transitioned to management systems that the ecological analysis identified as most efficient.

This dissertation is organized into four chapters: this introduction, and the three chapters reporting the findings of these three studies. The overarching theme that weaves these elements together is linking food production, ecology and humans. My work spanned a very broad continuum of farming types in the upper Midwest, particularly seeking to include understudied alternative systems, and to document both their management and their socioecological outcomes. In my work I sought to understand how different farming systems impact environmental sustainability, and how biophysical, socioeconomic and political factors shape opportunities for widespread transformation to more ecologically efficient systems.

Chapter 1: Comparing nitrogen management practices using a mass balance approach

The first study explored the relationship between a broad gradient of agricultural management practices on working grain farms in the upper Mississippi River Basin and their potential for nitrogen loss. Industrial grain agriculture is characterized by numerous ecological disconnections, and the root cause of nitrogen leakiness is the uncoupling of carbon and nitrogen biogeochemical cycles. In this work I built upon a quantitative synthesis I had conducted of over 200 short-term and small-scale studies using stable isotope methods in temperate grain agroecosystems (Gardner and Drinkwater, 2009). That synthesis indicated that management practices which add carbon and nitrogen together—either by increasing the complexity of crop rotations or by applying nitrogen in organic forms—have greater potential to improve nitrogen retention than practices that focus on adjusting the precision of commercial nitrogen fertilizer inputs. However, most of this evidence comes from short-term and small scale studies that fail to capture a full crop rotation, or the diversity of ways in which farmers actually manage their land, given the reality of environmental and social variability. Few data exist from working farms and this novel on-farm data set contributes to filling that gap.

This study applied ecological theory to agricultural nutrient management. Agroecology is the application of ecological science to agriculture (Lowrance et al. 1984) and the ecosystem concept emphasizes interrelationships among organisms and their environment (Tansley, 1935). Ecological knowledge has been applied to development of cropping systems that are based on replacing external inputs with management of ecological processes and biotic interactions (Shennan, 2008). This requires targeting ecological processes across levels of organization from organisms to ecosystems. For example, the selection of crop species and varieties impacts population level processes, or plant diversity can be manipulated (e.g., through complex rotation sequences, cover cropping, or intercropping) to impact weed competition, pest and disease suppression, and soil fertility and nutrient cycling (Jackson et al. 2007; Drinkwater et al. 2008). Processes occurring at one spatial or temporal scale have impacts at other scales, and ecological management involves dealing with complexity, uncertainty, and managing disturbance (Shennan, 2008). Ecological metrics such as net primary productivity, energy use, biodiversity indices, or mass balances are useful for assessing the ecological efficiency of contrasting agricultural systems.

We used an ecological mass balance approach to assess the nitrogen use efficiency of a broad range of management practices on grain farms in the upper Midwest. Nutrient budgets have been used to understand nutrient cycling across the full range of spatial scales, and they are a useful metric for comparing the sustainability of different management practices. Because surplus nitrogen is a driver of nitrogen leakiness in all ecosystems, it is an important number to understand. We tested the hypothesis that coupling carbon and nitrogen cycles would lead to reduced nitrogen surpluses and potential for nitrogen losses on working farms. The goal of this work was to contribute to our understanding of how to achieve a better balance between nitrogen inputs and exports from grain agroecosystems, and, more broadly, to balance sufficient food production with restored water quality and sustainable food production.

Chapter 2: Exploring how farmers transition to ecological management in the upper Midwest

As the research for the first paper progressed, I became increasingly concerned with the question: why isn’t ecological knowledge applied to most current agricultural landscapes in the U.S.? That is, as my understanding of agroecological science deepened, so too did my understanding that there is a vast body of existing knowledge that is not currently reflected in policy or practice. Critical social science disciplines, such as political ecology, argue that: “environmental problems cannot be understood in isolation from the political and economic contexts within which they occur” (Bryant and Bailey, 1997, p. 28). There is a need to incorporate various constructivist epistemologies based on the understanding that what can be known is situated within a social, political and historical context (Turner and Robbins, 2008). Analyzing and critiquing societies’ self-understandings is necessary for explaining socioecological phenomena (Sayer, 1992). This qualitative study of 18 farmers who transitioned to the most ecologically efficient practices on grain farms in the region—which are incredibly rare on the landscape—incorporates epistemologies and methodologies typically marginalized in natural science inquiry.

My goals in bridging disciplines, and, in particular, selecting qualitative social science methods, were multifaceted. First, I aimed to fill a gap in the use of interpretivist methods on the larger research collaboration, which I hoped would also further my own ability to effectively collaborate on diverse interdisciplinary research teams in the future. The second goal was to gain deeper insight into social processes in agrifood systems that are not easily explained with quantitative methods or predictive models of cause and effect, recognizing that social science must diverge from natural science methods by including interpretive understanding of meaning. Third, through my position as an ecologist engaging in this work, I hoped to create new spaces within natural science communities of practice for social science methods to be valued as legitimate for gaining useful and systematic knowledge about the world. And a final goal was to support transitions to ecologically sustainable and socially just food systems more broadly—by providing material for academic theorizing about farm-level transition processes, as well as practical knowledge to serve social movements, organizations, policymakers, farmers and other actors engaged in constructing alternative agrifood systems at multiple levels of organization. This project thus reflects a philosophical pragmatism—by aiming to create knowledge to make a change—and the ontology of critical realism, which recognizes scientific knowledge as partial and socially constructed (Bhaskar, 1997; Sayer, 1997). That is, my work paid careful attention to the material and ecological world while simultaneously recognizing the discursive nature of scientific knowledge.

In this study, I compared two contrasting regions of Iowa. This helped me understand both the social and biophysical components of clustering—the phenomenon whereby innovation occurs in clusters where there are enabling resources, organizations, policies, and incentives. In the hilly region of northeastern Iowa, there is a cluster of organic farmers and rotational graziers in close proximity to the headquarters of Organic Valley. This sharply contrasts with the very rare occurrence of such alternative practices in the central Des Moines lobe region, which is dominated by industrial corn and soybean production. My personal research experiences reflected these differences among the two regions. Finding alternative farmers to participate in the study was far more challenging in the Des Moines lobe region compared to the Northeastern region, requiring many more contacts and 'leads.' Further, there were multiple instances where I scheduled an interview with a farmer whose address was in the Des Moines lobe region, but arrived to discover that their land was a small or isolated parcel that was hilly, or sandy, or ecologically marginal in some other way. At first I found this frustrating, considering it to be a research flaw since I was seeking to control for large variability in the background biophysical environment across farms. Over time, though, as I became familiar with the region, I understood that these actually existing arrangements reflect the dynamic convergence of history, ecology and politics on this particular landscape, where the industrial agricultural system has exploited nearly every tract of prime farmland for the purpose of capital accumulation. Given this particular socioecological context and ordering, it is unsurprising that the agricultural practices at the most ecological end of my management gradient would be extremely rare in this central region.

In the two contrasting regions, I explored farmer innovation dynamics in a systemic and relational way by elucidating micro-level factors such as the knowledge, skills, values, biophysical and ecological conditions on farms, and material and social resources, which enabled farmer transitions to occur. I viewed farm-level knowledge and resources as being nested within both ecological and social relations of production, and those, in turn, as nested within the macro structural context of the neoliberalization of agriculture. This study contributes both a multi-level and relational analysis, as well as an integrated socio-ecological perspective grounded in ecological data collected from farms, to the study of innovation processes that lead to environmental conservation and sustainability.

Chapter 3: Winter annual cover crops and the cycling and retention of ^{15}*N-labeled fertilizer in an Illinois prairie soil*

As a dissertation in the fields of soil science and agroecology, I wanted to include a mechanistic study to contribute to a basic understanding of nitrogen cycling processes on grain farms. The mass balance study indicated that ecological practices which re-couple carbon and nitrogen cycles were the most efficient in terms of reducing potential for nitrogen pollution. In this study, we investigated one specific ecological practice that was of particular interest to the larger research team—including winter annual cover crops in rotation to reduce the length of time grain fields are in a bare fallow, which is the most problematic period of nitrogen losses from farms in the region. Studies that measure nitrate leaching have shown cover cropping to be a promising practice; however, very few studies have been conducted on cover cropping in the Corn Belt. We sought to understand how effective winter cover would be at reducing nitrogen pollution in Illinois, which is one of the leakiest states in the upper Midwest (David et al. 2010). We used stable isotope methods in a corn-soybean rotation on an Illinois prairie soil to quantify differences in recovery of recently added ^{15}N-labeled nitrogen with and without winter cover and to elucidate the biogeochemical mechanisms by which cover crops can sequester nitrogen. To do that, we measured ^{15}N recovery in a wide range of soil organic matter pools with differing turnover times. This makes a key contribution to a body of literature that has primarily focused on the single pathway of fertilizer application, soil inorganic nitrogen pools, and crop nitrogen uptake. This project thus aimed to deepen our understanding of the potential for intrinsic ecological processes to improve N use efficiency.

Unfortunately, very high rainfall in October during the year of the experiment meant that we were not able to plant the cover crop in time to catch the optimal window for capturing soluble fall nitrogen. This meant that what was initially a tracer study intended to explore mechanisms by which cover crops sequester nitrogen in soils, became more of a study on the pathways of nitrogen cycling through heterogeneous soil organic matter pools. Spring cover crop growth allowed us to draw conclusions about the role of cover crops in scavenging nitrogen mineralizing from soil organic matter with spring soil warming, but not the specific mechanisms of sequestering newly added fertilizer, since much of the soluble labeled nitrogen was lost with rainfall before we imposed the treatments of cover and bare fallow. The study, however, allowed us to draw conclusions about the pathways of ^{15}N recovery in a wide range of soil organic matter pools in relation to corn yields and temporal dynamics of fertilizer cycling from fall to spring.

Further, the difficulties we experienced in establishing a cover crop in Illinois perhaps reflect the challenges actually faced by farmers in this region in integrating winter cover into industrialized crop rotations. There is need for more research to develop grain varieties and cropping systems that are optimized to include ecological practices that hold the most promise for nutrient cycling efficiency while maintaining productivity.

Taken together, this work contributes an integrated perspective on social and ecological factors that drive nitrogen leakiness—or retention—in the region which contributes the bulk of the nitrogen load that causes hypoxia in the Gulf of Mexico. Given the complexity of the environmental problems of industrial agriculture, I would argue that this type of systemic approach holds the most promise for understanding barriers and opportunities to innovation toward agrifood systems that improve environmental quality and social justice. Integrated, interdisciplinary research spans multiple epistemological perspectives. Researchers employ methods in particular ways based on our ontological and epistemological assumptions, which have very real downstream consequences. All scholars, including natural scientists, have much to gain from reflecting on how our worldviews drive the research process and resulting outcomes. Ethical thinking is an important part of science. Further, social science must also include a normative component and a critical perspective that questions our beliefs about reality; when we reflect upon these beliefs they often change. Hence there is always an interpretive dimension to social science that is distinct from natural science methods. For instance, questioning power relations is of the utmost importance: Who benefits from agricultural systems and technologies in the Mississippi River Basin? And, who gets to decide which technologies will be supported and advanced?

A growing body of research is exploring the interdisciplinary research process itself, seeking to understand the philosophies and conditions that lead cross-disciplinary collaborations to succeed, or not. Interdisciplinary teams frequently come together and converse, but then retreat back into disciplinary silos without successfully producing an integrated product. One potential solution is a placing greater educational emphasis on training interdisciplinary individuals, and developing institutional reward structures that support their work. In this dissertation, I aimed to practice interdisciplinary work to make a contribution to development of sustainable agrifood systems, which meant incorporating multiple disciplinary and epistemological perspectives without reducing one to another—I linked biophysical science with critical social science theories and methods. Bridging multiple literatures is a common activity for doctoral students, but it is extremely challenging to bring such diverse perspectives into effective conversation with one another. Why take on the challenge of integrating diverse perspectives and traditions in an academic culture that has privileged the development of highly specialized and compartmentalized knowledge? Because the barriers to sustainability are not simply technical, they are equally social and political, and research that contributes to overcoming these barriers will require new theories and approaches that are engaged in communities.

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CHAPTER 1

COMPARING NITROGEN MANAGEMENT PRACTICES ON GRAIN FARMS IN THE MISSISSIPPI RIVER BASIN USING A MASS BALANCE APPROACH

Abstract

Nitrogen (N) leaching to surface waters from grain farms in the Mississippi River Basin (MRB) is the primary cause of hypoxia in the Gulf of Mexico. Regional-scale N mass balances indicate that a small area of the upper MRB, which is intensively cropped and tile-drained, contributes disproportionately to nitrate loading. These aggregate balances miss small-scale variability, especially that caused by differences in farm management. A better understanding of the flows and fates of N at the field level is needed as an indicator of directional change towards deficit or surplus (i.e., potential for N loss) and to compare the relative efficiency of diverse farming systems in the MRB. We constructed N mass balances for a gradient of farm types from intensive corn-soybean monocultures to diversified grain farms that rely on biological N fixation (BNF) as a primary N source. Five-year N balances were calculated for a most- and least-productive field on each farm using data on major field-scale N fluxes collected from interviews with 95 grain farmers conducted between 2007-2009 on grain farms in Iowa, Ohio, Minnesota and Wisconsin; from legume biomass and corn grain samples collected from a subset of farms; and published values from the literature. Spring and summer legume biomass, and summer measurements of legume reliance on BNF were highly variable, demonstrating the importance of using sampling data to improve estimates of uncertain N fluxes for working farms. Nitrogen balances ranged from high average annual surpluses (149 kg N ha^{-1}yr^{-1}) to large deficits (80 kg N ha^{-1}yr^{-1}), and differed based on N source and crop rotation. Fields with greater than 50% of total

N additions from legume N sources, and fields with complex crop rotations that included both annual and perennial species, were approximately in balance, compared to fertilizer-based practices in corn-soybean rotations with average annual surpluses near 35 kg N ha^{-1}. Surplus N was also inversely related to the proportion of total N inputs from BNF for medium (80-160 kg N ha^{-1} yr^{-1}) to high (>160 kg N ha^{-1} yr^{-1}) N rates. Finally, we found that diversified farmers were more likely to adjust their management practices in response to environmental variability compared to fertilizer-based farmers. In aggregate, farmers who put less N on their least productive field had smaller N surpluses than farmers who put more N on their least productive field. Together, results from this study suggest that significantly reducing surplus N in agroecosystems will require reducing N inputs and increasing C availability (i.e., coupling C and N cycles) to support the internal biological mechanisms for storing N in farm fields.

Introduction

The consequences of how food is produced extend far beyond the boundaries of individual farms. A prime example is nitrogen (N) pollution from grain farming in the upper Midwest, which causes a hypoxic zone to form in the Gulf of Mexico each summer thousands of miles from the origin of N leaching losses (Carpenter et al. 1998; McIsaac et al., 2001). Agriculture has become the leading source of water pollution in the U.S., and there are more than 400 scientifically-defined hypoxic zones in coastal marine ecosystems worldwide (Diaz and Rosenberg, 2008). Over the past several decades, growing recognition of the large social and ecological costs of resource-extractive, industrial agriculture (Matson et al. 1997; MEA, 2005) has been paralleled by growing calls for new approaches to food production that sustain both ecosystems and people (IAASTD, 2008). Addressing complex, systemic environmental crises such as water pollution, climate change, biodiversity loss, and soil erosion, will require interdisciplinary research, as well as communication and cooperation among diverse actors inside and outside of academia. Given these current conditions, the research we present here centers on several goals: to help advance the application of ecological knowledge to agricultural research and practice, to conduct on-farm work that seeks to understand agriculture in its 'actually existing' context, and, therefore, to exemplify agroecological science that is engaged and situated, which we argue is more likely to successfully address today's confluence of crises. This research is part of a larger interdisciplinary collaboration exploring linkages between human and natural processes in agriculture, seeking to understand how complex, socio-ecological systems co-evolve towards or away from sustainability (Norgaard, 1994). To do that, the collaborative work focused on agricultural landscapes of the Mississippi-Atchafalaya River Basin (MRB) and the hypoxia problem in the Gulf of Mexico as a case study.

Industrial agriculture accounts for the greatest proportion of anthropogenic N forcing globally (Galloway and Cowling, 2002), primarily with increasing applications of inorganic N fertilizers since N frequently limits crop productivity (Vitousek et al. 1997). Reactive N production by humans has increased by 120% since 1970 (Galloway et al., 2008). Such modifications of the global N cycle have uncoupled carbon (C) and N biogeochemical cycling in industrial agroecosystems, leading to their leakiness (Woodmansee, 1984; Drinkwater and Snapp, 2007; Gardner and Drinkwater, 2009). In natural ecosystems, microbial processes such as decomposition, N fixation, and denitrification drive C and N cycles because microorganisms require both C and N for respiration, maintenance, and growth. In fertilizer-based agroecosystems—which have smaller soil organic matter (SOM) pools, and reduced periods of living plant cover, C fixation and C additions than most natural ecosystems—microbial assimilation of inorganic N fertilizers is typically limited by the availability of C (Drinkwater and Snapp, 2007). As a result, in these systems, pulse additions of fertilizer N saturate the plant and microbial C-sinks for N, reducing the potential for N assimilation and N use efficiency. Further, the reliance on inorganic N forms exacerbates N leakiness since these soluble forms cycle rapidly and are vulnerable to loss through multiple pathways.

The MRB spans approximately 40% of the land area in the conterminous U.S., and is predominately agricultural lands. Nitrate leaching losses from a relatively small area of approximately 100 counties in the Corn Belt of the upper MRB contributes the greatest proportion of total N delivered to the Gulf of Mexico (David et al. 2010). The farming systems in the 'leakiest' counties are dominated by alternating monocultures of corn (*Zea mays* L.) and soybeans [*Glycine max* (L.) Merr.]. The systemic intensification and industrialization of agriculture in the region eliminated plant species from the landscape that used to contribute to maintaining ecological functions, such as legume cover crops and perennial forages, and replaced them with chemical inputs like N fertilizer. Corn and soybeans comprise up to 90-95% of the land area in some of the most intensively farmed counties in the basin (David et al. 2010) and winter bare fallow periods (i.e., when no living plants are in agricultural fields following harvest of spring annuals) typically last 4-8 months. In addition, agricultural production from the high organic matter, former prairie soils of the region was augmented by installation of artificial subsurface drainage (tile drainage) in farm fields. Tile drains intercept natural water flow to groundwater and speed up water movement from the landscape by diverting it to surface waters (Dinnes et al. 2002). Increasing tile drainage in farm fields has likely enhanced the importance of leaching as a loss pathway of the N cycle in this region.

Several lines of evidence at field and farm scales suggest that agronomic practices which manage C and N together reduce N surpluses and improve agroecosystem N retention. First, this hypothesis is supported by two meta-analyses of the agroecology literature, which summarized short-term and small-scale experiments. The first meta-analysis assessed cover cropping—a management practice that re-couples C and N cycles but is not commonly used in fertilizer-intensive grain systems of the MRB. The authors reported a 70% reduction in nitrate leaching when non-legume cover crops replaced a bare fallow period in intensive cash grain systems, without affecting crop yield. Rotations with legume N sources and winter cover averaged 40% less nitrate leaching (Tonitto et al. 2006). In the second meta-analysis, we expanded on this approach to investigate a more comprehensive range of management practices (Gardner and Drinkwater, 2009). Our analysis summarized over 200 experiments using ^{15}N tracer methods to follow the fate of N in grain cropping systems with a variety of improved management strategies. We grouped the practices for analysis based on the theoretical framework underlying the particular approach to soil fertility management. The fertilizer-based practices included adjustments to inorganic fertilizer applications such different chemical forms, methods, timing, or reduced rates. A second category of experiments investigated management practices that target multiple processes of the N cycle (for example, N mineralization *and* crop uptake) and re-couple C and N cycles at multiple scales. The two ecological practices—diversified crop rotations and organic N sources—significantly improved total N retention compared to common fertilizer-based strategies, including reduced N rates, nitrification inhibitors, and changing chemical forms of fertilizer (Gardner and Drinkwater, 2009).

Second, N mass balances calculated for cropping systems experiments have demonstrated that more diverse cropping systems (i.e., including cover crops or winter grains in rotation to reduce bare fallow periods) that rely on legume N sources rather than commercial fertilizer maintain yields and reduce N losses (Drinkwater et al. 1998; Gregorich et al. 2001; Ross et al. 2008). In part, the improved efficiency is derived from reductions in surplus N additions and the use of more stable, organic N sources that are mineralized through microbial processes. These experiments are long-term comparisons of distinct, intact management systems, such as organic, low-input, or conventional. Though only in a small number of sites, the strength of these complex experiments is that they are long-term systemic comparisons, which complement more typical reductionist assessments of single agronomic factors.

Mass balance approaches to studying biogeochemical cycles of elements have become widespread since their development in ecosystem ecology several decades ago (Scoones and Toulmin, 1998). Input-output balances for agroecosystems identify deficits or surpluses of nutrients and, therefore, the potential for pollution. Because surplus N is a major driver of N loss in all ecosystems, even in forests (Fenn et al. 1998; Drinkwater and Snapp, 2007), it is an important number to understand. While the N balance data from cropping systems experiments is promising, mass balance data from working farms is scarce due to the difficulty of controlling for variability. Yet testing these hypotheses on working farms is important because data from cropping systems trials are collected at a single location and fail to capture the effect of varying environmental and social conditions on the responses of interest. In addition, in trials at agricultural experiment stations, a given management approach, such as ‘organic,’ is represented by one set of practices. This decision lends itself to statistical analysis, but the disadvantage is that the experiment does not capture, or test, the diverse ways in which farmers manage their land given the current environmental, socioeconomic and political context (Drinkwater, 2002). In addition to the lack of on-farm data, most published N balances for the upper MRB are aggregated based on geopolitical units (i.e., because this is the data that is available), and thus do not capture farm- and field-scale variation due to environment and management. However, a better understanding of the flows and fates of N at the field level is important as an indicator of directional change towards deficit or surplus, and for comparing the relative efficiency and sustainability of a variety of management approaches within a region (Drinkwater et al. 2008). In particular, there is a need for research that characterizes nutrient flows in understudied alternatively managed systems.

To address this knowledge gap for farms in the upper MRB, we tested the hypothesis that management practices that increase C-N coupling would result in reduced N surplus and N loss potential across a wide gradient of grain farm types, spanning continuous corn fields to highly diversified organic grain farms. We also predicted that reliance on legume N sources would be inversely related to N surplus because legumes fix both C and N, and are capable of responding to environmental conditions and regulating biological N fixation (BNF) through internal feedback mechanisms (Schipanski et al. 2010). This research addressed two broad questions about management and environment: Do ecological nutrient management practices that increase C-N coupling result in reduced N surplus and N loss potential across a gradient of farm types in the MRB? And, do farmers adjust their management practices in response to environmental variability across different farm fields?

Methods

Research Sites and Selection of Farmer Participants

To address these questions on grain farms in the upper MRB, we selected farmer participants (n=95) from four study sites (Figure 1.1): the Des Moines lobe (DML) in central

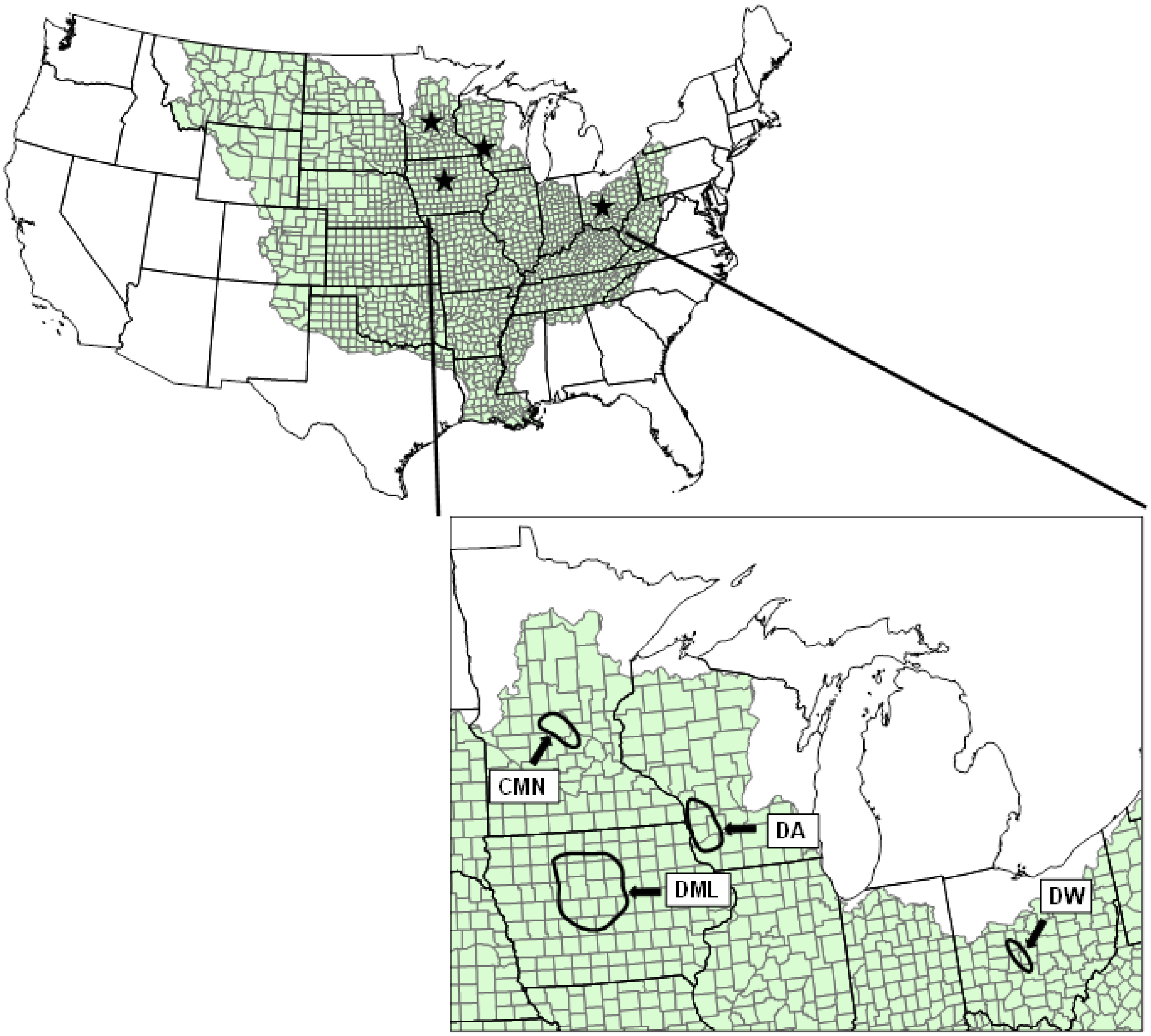

Figure 1.1. U.S. map highlighting counties within the Mississippi-Atchafalaya River Basin (MRB). The four study sites are marked with black stars. The inset defines the boundaries of the study sites more clearly with their abbreviations: the Des Moines lobe (DML) in central Iowa, the Darby Watershed (DW) in Ohio, Stearns and Wright counties in central Minnesota (CMN), and the Driftless Area (DA) in southwestern Wisconsin.

Iowa, the Darby Watershed (DW) in Ohio, Stearns and Wright counties in central Minnesota (CMN), and the Driftless Area (DA) in southwestern Wisconsin. These sites represent geopolitical units (e.g., counties or clusters of counties) or biophysical units such as watersheds or soil regions. We balanced two criteria during site selection: i) including regions of the Midwest known to contribute substantial N pollution to the Gulf of Mexico and, ii) including a broad range of industrial and alternative nutrient management practices. The region is characterized by a humid continental climate with mean annual precipitation values ranging from 823 - 978 mm, with the exception of the drier Minnesota site which has a mean annual precipitation of 688mm. The mean annual maximum air temperatures across the sites range from 11 - 17°C and mean annual minimum air temperatures range from 0 - 6°C.

Within each of these sites farmers were purposefully selected to capture the full range of management practices found on grain farms in the Midwest. Therefore, farmers in this study do not comprise a representative sample of all grain farmers in the MRB; instead, the strength of this sampling approach is that it allows us to draw conclusions about the relationship between the full spectrum of management types in the region and their consequences for potential N losses. The management gradient spanned practices from industrial continuous corn cropping with heavy reliance on external chemical inputs to highly diversified, certified organic grain farms that rely on legume N fixation as the primary N source. As a result, total farm size varied across participants—ranging from 48 to 2548 hectares (mean of 480 ha)—however, to reduce potential variability in farm size we did not include the smallest or largest farms found in this region in the sample. We contacted individuals from a large number of organizations in each site to connect us with farmer participants including cooperative extension, commodity groups, organic certifying agencies, and farmer organizations; academics who conduct research with farmers; and staff in county Natural Resources Conservation Service field offices and state Department of Natural Resources offices. We also asked all farmers interviewed to suggest other farmers who would potentially be interested in participating.

Construction of N Mass Balance Approach

We combined data from farmer interviews, on-farm plant sampling, and assumptions from the literature to construct five-year, field-scale N mass balances for two fields on each of the 95 farms. Five-year N mass balances were calculated as the difference between total N inputs from commercial fertilizer, animal manure, and legume N fixation, and the total N exported in harvested grain, silage or straw (Figure 1.2). These are the largest, most important N flows controlled by farmers. Large N surpluses (i.e., positive mass balances) indicate the potential for N leaching losses and pollution. Because surplus N correlates with N losses for a wide range of ecosystem types (Drinkwater and Snapp, 2007) we focused on this metric as an indicator of N use efficiency and potential for pollution that is useful for comparing working farms. Deficits of N (i.e., negative mass balances) indicate potential degradation of SOM pools over time.

Farmer Interviews

The first author conducted interviews with all farmers between 2007 and 2009. Each interview was approximately one to two hours in length, and included farm-scale questions about land use, dominant soil types, crop rotations, topography and tile drainage. We also collected basic socioeconomic and demographic information including participation in federal conservation and commodity programs. The primary focus of the interviews, however, was to collect detailed information about farmers' crop rotations and N management practices in order to calculate N mass balances. During each interview, farmers were asked to select two fields that had been planted with a corn crop in the most recent growing season, and which had variable productivity (i.e., a most and least productive corn field), in order to capture a range of environmental conditions on each farm. For each of these two fields we collected basic information about field location, size, tillage practices, tile drainage, slope, soil types, and proximity to waterways.

We then asked detailed questions about N management for these two fields during the most recent corn year including fertilizer practices, legume or manure management, and corn variety and yield. The data sources and methods used to collect data for each of the N balance terms are summarized in Table 1.1. To calculate N inputs for the most and least productive fields we collected the following data from farmers about the most recent corn year:

i) Commercial N fertilizer: type of fertilizer, total N rate, timing, and application method

ii) Livestock manure: type of manure and nutrient analysis, application rate, date, and method

iii) Legume green manure or hay: legume species, date planted, seeding rate, date plant biomass was incorporated in soil, and incorporation method

Whenever possible we relied on farm records to help ensure accuracy. The total amount of livestock manure applied and nutrient analysis of the manure were used to estimate manure N inputs. If farmers were unable to supply a recent manure nutrient analysis, the manure N content was estimated using values in regional cooperative extension publications[1]. Nitrogen inputs from BNF by legume crops were estimated by combining information on legume species, biomass and management from the farmer interviews, with plant N concentration, and measurements of percent of aboveground plant N from fixation from plant sampling (see explanation in next

[1] We used the following sources to estimate animal manure N inputs when farmers weren't able to provide a recent nutrient analysis: i) Table 5 in Cooperative Extension Service. 2001. *Animal Manure as a Plant Nutrient Resource*. Purdue University, West Lafayette, IN., and ii) Table 2 in University Extension. 2003. *Managing Manure Nutrients for Crop Production*. Iowa State University, Ames, IA.

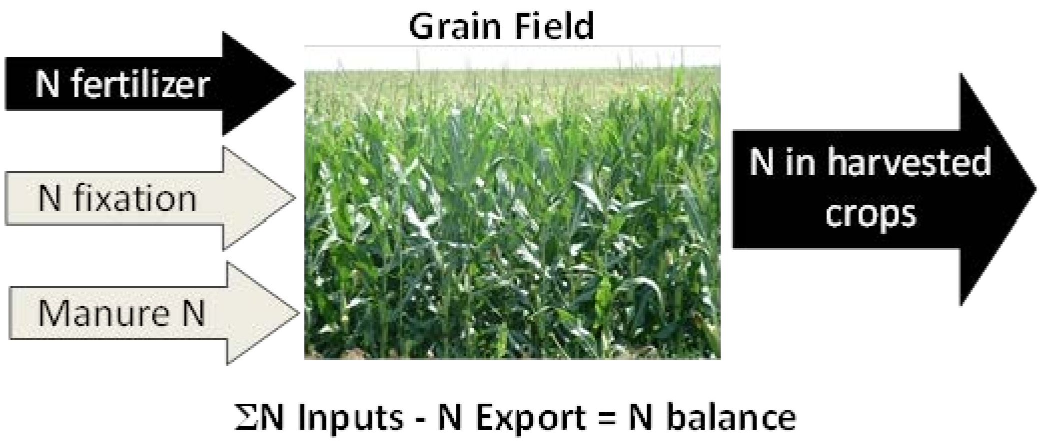

Figure 1.2. The field-scale mass balance approach used to calculate five year N balances for two fields per farm, which focused on the largest N fluxes controlled by farmers. Inputs were from commercial N fertilizer, legume N fixation, and animal manure, and exports were in harvested crops. The N balance is the sum of the N inputs minus N exports. A positive N balance represents surplus N, and a negative N balance indicates deficits of N. Fertilizer inputs and harvested N exports (black arrows) are the most certain terms in the balances. Quantifying N inputs from legume N fixation and animal manure (gray arrows) require assumptions from the literature and are less certain terms.

Table 1.1. Summary of data sources used to calculate the N mass balance parameters.

N Mass Balance Terms	Method/Source
N Inputs	
N fertilizer	
type/rate	farmer interviews
N content	literature
Livestock manure	
type/rate	farmer interviews
N content	farmer nutrient analysis or literature
Legumes -aboveground	
species/biomass	farmer interviews
N content	plant sampling
N harvest index	plant sampling
% N from fixation	NA method and literature
Legumes- belowground	
N content/ % N from fixation	literature
rhizodeposition/exudation	literature
Atmospheric deposition	NADP/NTN
N Exports	
Harvested crops or hay	
grain yield or hay yield	farmer interviews
corn N content	sampling
other crop or hay N content	NRCS Crop Nutrient Tool

section). We also estimated belowground legume N inputs due to root turnover and exudation and upon incorporation of the legume using values from the literature. To calculate N exports in the corn year we combined the yield data from the interview with a corn grain N concentration from on-farm samples (see next section).

After collecting this in-depth management data for the most recent corn year in each of the fields, we collected information on the crop rotation, total N inputs and yields for the four previous years, in order to calculate a five-year N mass balance for the most and least productive field on each farm. To calculate total harvested N exports from each field for crops other than corn, data on crop rotations and crop yields from the interviews were combined with average N concentrations listed in the Natural Resources Conservation Service Crop Nutrient Tool (USDA, 2009), an online database for estimating nutrient removal from agricultural fields in crop harvests. Because soybeans are also legumes, to calculate the N balance in soybean years we combined data on soybean yield collected from the farmer interviews with published values from the literature. We assumed that 57% of aboveground soybean N was from fixation (David and Gentry, 2000; Russelle and Birr, 2004; David et al. 2010); that soybeans had an 80% N harvest index (David and Gentry, 2000), and that the beans were 6.4% N (USDA, 2009). In addition, we conducted a sensitivity analysis varying the soybean N fixation rates (from a low of 50% to a high of 80%) since this N flux is difficult to precisely quantify and because this assumption potentially has large impacts on the mass balance results due to the frequent occurrence of soybeans in Midwestern crop rotations.

Plant Sampling

The N mass balances, which could have been calculated using the interview data combined with literature values, were improved with field sampling. Grain N removal and BNF are known to vary with management (Ma et al. 2006; Oberson et al. 2007), and represent important fluxes in farm N balances; we therefore used sampling to improve our estimates of these values in the mass balance calculations.

Corn Samples

We collected corn grain samples from farmers by mail in fall of 2008 and fall of 2009 from 61 and 68 farms, respectively. Briefly, the corn grain N method involved having farmers mail us a small, representative grain sample (approximately 250g) from their grain bin as they were harvesting one field. They also returned a data sheet with the sample listing the corn variety, yield, and field size. Samples were dried at 60°C, ground using a hammer mill and grinder, and total N concentration was measured using a Leco 2000 CN Analyzer (Leco Corporation, St. Joseph, MO). The resulting N concentrations were entered into the mass balance database for each specific farm. We estimated harvested N exports for farms from which we were not able to collect corn samples using the %N of corn samples from farms in the same study site, with the same or similar corn varieties and management systems.

Legume Biomass

To make more robust estimates of N inputs from BNF, we sampled legume aboveground biomass on a subset of the farms that had legume species in rotation, which were selected using a nested sampling design. Other than soybean, the dominant legume species on the farm sites were alfalfa (*Medicago sativa*) and red clover (*Trifolium pratense*), and a small number of fields had white clover (*Trifolium repens*). Samples were collected in summer 2008, spring 2009, and summer 2009. The summer samples (n = 30 fields total) focused on perennial hay fields and captured the window just before second and third hay cuttings, and spanned stand ages of one to four years. The spring sample (n = 14 fields) provided an estimate of spring legume growth just prior to incorporation by tillage and planting of a grain crop. Legume biomass was harvested at the soil surface from 0.25 m^2 plots (4-6 replicates per field). Biomass was dried at 60°C, ground using a hammer mill and grinder, weighed, and analyzed for total C and N on a Leco 2000 CN Analyzer (Leco Corporation, St. Joseph, MO). If the stands were mixed grass-legume, samples were separated into individual legume species and grasses. On a subset of hay fields (n = 18) we simulated haying with our sampling (i.e., cutting the legumes first to the farmer's haying height and then cutting the remaining stems and biomass) in order to calculate an aboveground N harvest index for hay fields.

Estimates of N fixation using ^{15}N Natural Abundance

In the 2009 field season we selected a subset of farms that were sampled in 2008 and 2009 to directly measure BNF inputs using the ^{15}N natural abundance method (Shearer and Kohl, 1986). The natural abundance method relies on small differences in the δ^{15}N signature of legumes and non-legumes, since legume N comes from two sources—soil and the atmosphere. The method requires that the biological variability of δ^{15}N abundances in either pool is small compared to the differences in isotopic signatures between the two pools (Boddey et al., 2000). We sampled 10 fields immediately before the second hay cutting to measure the δ^{15}N of alfalfa and red clover, which were the dominant legume species in organic rotations in the MRB. We simultaneously sampled one to two adjacent reference species (non-legumes assumed to be accessing the same pools of soil N as the legume). The most common reference plants were orchardgrass (*Dactylis glomerata* L.) and timothy (*Phleum Pratense* L.). Legume and reference plant biomass was harvested at the soil surface for 4-6 replicates per field. All samples were dried at 60°C and weighed. The dried samples were coarsely ground using both a hammer mill and grinder, and then pulverized to a very fine powder using a roller grinder. Samples were analyzed for total N content and ^{15}N natural abundance on a continuous flow Isotope Ratio Mass Spectrometer (Stable Isotope Facility, UC Davis). We calculated %N derived from fixation using the following mixing model: %N from fixation = $100*((\delta^{15}N_{ref} - \delta^{15}N_{legume}) / (\delta^{15}N_{ref} - B))$, where $\delta^{15}N_{ref}$ is the $\delta^{15}N$ signature of the reference plant, $\delta^{15}N_{legume}$ is the $\delta^{15}N$ signature of the legume, and *B* is defined as the $\delta^{15}N$ signature of a legume when dependent solely on atmospheric N_2. The *B* value quantifies the ^{15}N fractionation that occurs during BNF for each species, which is affected by both rhizobial strains and internal translocation of N from roots to shoots.

The *B* values were determined by growing two varieties of alfalfa and three varieties of red clover, which had been sampled in the field, in a growth chamber in a N-free medium. Seeds were surface sterilized in 70% (*v*/*v*) ethanol for three minutes and rinsed three times with deionized water. Seeds were then soaked for an additional three minutes in 3% (*v*/*v*) NaOCl and rinsed three more times with deionized water. The seeds were coated with 1g of the recommended Nitragin® Gold inoculant (i.e., clover or alfalfa). Five replicates of each variety were planted (16 seeds/pot, thinned to four plants per pot) in a N-free, autoclaved perlite/sand media (1:5 perlite:sand) in pots that had been soaked in 3% NaOCl. Pots were arranged in a randomized complete block design in a chamber with 16h day length, a daytime temperature of 25°C, and nighttime temperature of 15°C. Plants were watered with deionized water, and were fertilized with a N-free nutrient solution. Whole plants were harvested when almost all plants were flowering, separated into roots and shoots, dried, finely ground, and analyzed for $\delta^{15}N$. We used the following B values in our calculations: -0.73 for Medium red clover varieties, -1.0 for Mammoth red clover, and -0.7 for alfalfa. This isotopic work was used to improve the accuracy of the field-scale mass balances by quantifying BNF rates across the range of management types.

Modeling Legume N Inputs

The subset of measured BNF rates were used to predict BNF rates for all fields with alfalfa and red clover in the database. Modeling total N fixation inputs over the 5-year crop rotation was a multi-step process. First, because we measured N fixation rates at their summer peak, we adjusted the mean rate from our sampling distribution to a more accurate rate for the entire growing season using data from Kelner at al. (1997) who measured alfalfa N fixation rates across the entire growing season for multiple years. The values for %N from fixation used in our database were: 76% for red clover, 68% for alfalfa in the seeding year and 70% for alfalfa in all other years. Because of high variability in our measured rates of alfalfa N fixation across fields, we also conducted a sensitivity analysis varying BNF rates using the 25% and 75% quartiles for each distribution. Thus, in this analysis, alfalfa BNF ranged from a low of 45%N from fixation to a high of 87%, and red clover rates ranged from a low of 70% to a high of 84%N from fixation.

Legume aboveground biomass was estimated from farmer interviews (i.e., biomass removed at each hay harvest), and we combined this number with the aboveground N harvest index (NHI) and N content (%N) from our simulated 'hay' sampling. We estimated that belowground N inputs from rhizodeposition were 1.5 - 5% of total aboveground fixed N yr^{-1} depending on the stand age (Ta et al. 1986, Lory et al. 1992). In the year of legume incorporation we estimated fixed N inputs from root biomass using the following assumptions: that 35% of total plant N is in the root biomass for both alfalfa and red clover (Reiter et al. 2000; Carlsson and Huss-Danell, 2003; Ross et al. 2008) and that 60% of root N is from BNF (Lory et al. 1992).

Some farmers incorporated their hay stands in the fall before planting a spring grain crop, in which case we did not add any additional N in the cash crop year. For farmers who incorporated their hay stands just before planting, we estimated additional N inputs from spring growth (i.e. prior to tillage) using the biomass and %N data from our analysis of spring legume samples.

Atmospheric Deposition

The primary focus of this study was exploring the relationship between farmers' nutrient management practices and N balance. However, anthropogenic N inputs from atmospheric deposition (NO_y) are also an important N source to farm fields, even if this input is not controlled by farmers. We therefore estimated these N additions for each study site. For wet atmospheric deposition, we used the annual NO_3^--N averages from the National Atmospheric Deposition Program/National Trends Network (NADP/NTN) sites closest to each study site (NADP, 2011). NH_4^+-N was considered an internal flux and not a net input (Howarth et al. 1996; McIsaac et al. 2002). We estimated dry deposition of HNO_3 and NO_3^--N as 70% of wet deposition following Goolsby et al. (1999) and David and Gentry (2000). Our mass balance approach also assumes that N volatilization losses from manure, and re-deposition of reduced N, is approximately balanced at the field-scale and represents internal recycling of N.

Statistical Analysis

We conducted all statistical analyses using JMP v.8 software (SAS Institute Inc., Cary, NC). We used mixed model analysis of variance (ANOVA) to test the mean separations for different N sources and crop rotations, with fields as fixed effects nested within farm, a random effect. Farmer nutrient management practices were separated into three N source levels for analysis: fertilizer-based, manure-based and legume-based. The fertilizer-based practices were surface broadcast applications of fertilizer, injected anhydrous ammonia, and sidedress or split fertilizer application. For the surface broadcast practice, fertilizer was spread or sprayed on the soil surface, in almost all cases immediately before planting or at planting. The most common fertilizer form for broadcast applications was urea ammonium nitrate (i.e., UAN; either 28 or 32%N). The anhydrous ammonia applications primarily occurred in the spring (two farmers applied the full amount in the fall) and were injected, or "knifed-in" to the row. The sidedress category included spring split-applications of fertilizer (multiple forms) where at least one application occurred post-emergence. We defined manure-based fields as having 50% or more of their total annual N inputs from manure N sources. The manure management practices were separated into either fall/winter or spring applications of manure. The legume-based fields were defined as fields with 50% or more of total N inputs coming from BNF.

We also defined three crop rotation categories based on crop functional group and frequency of corn in rotation, since corn is a large driver of the N balances in this region. The first category included fields that were either in continuous corn or corn-soybean rotations (i.e., only summer annuals, with corn appearing at least every other year). The second category included farms that had summer annuals with corn appearing less frequently than every other year—for example, a small grain such as spring wheat might be included in rotation—or, fields with both summer and winter annuals (e.g., winter wheat). The third rotation category included fields with both annual and perennial crop species in rotation. In the results section, references to *diversified farms* include all farmers in either the legume-based category or in the crop rotation category with both annuals and perennials. We avoid calling this group of farms *organic* because while all organic farms in this study fall into the *diversified* category, there were instances of diversified and legume-based farmers who were not certified organic. The practices analyzed in this study were intentionally selected to span a continuum of nutrient management practices rather than to fall into coarse categories like organic and conventional, each of which encompasses a wide variety of management approaches and captures differences in ecosystem structure imposed by management. For the analysis of most and least productive fields, farms were nested within study sites to account for potential differences in environmental variability across sites. We assessed all variables for normality and homogeneity of variance to meet the assumptions of ANOVA. We calculated post hoc pairwise comparisons using Tukey's HSD and pairwise comparisons using Student's *t*-tests. Categorical data were analyzed using chi-square goodness-of-fit tests.

Results

Corn Year N Mass Balance

For the fertilizer-based farms, N inputs ranged from 33 to 240 kg N ha^{-1} yr^{-1}, and the mean total N input on the fertilizer-based farms in the corn year of the rotation (i.e., the year in which all of the fields in the database were planted to corn) was 182 ± 4.9 kg N ha^{-1} yr^{-1} (mean ± s.e.m.). The N inputs on the *diversified* farms (i.e., which included all of the fields that had either complex crop rotations with annuals and perennials or greater than 50% of total N inputs from legume sources) spanned 0 to 398 kg N ha^{-1} yr^{-1}, with a significantly lower mean N input in the corn year (138 ± 11.6 kg N ha^{-1} yr^{-1}; $p < 0.0001$). Corn yields across all farms ranged from 0 – 14 Mg ha^{-1}. Corn yields were significantly greater in the fertilizer-based farms (mean = 9.7 Mg ha^{-1}; $p < 0.0001$) than in the diversified farms (mean = 6.1 Mg ha^{-1}). Harvested N exports in the fertilizer-based and diversified farms were not significantly different (91.2 ± 9.3 kg N ha^{-1} yr^{-1} and 99.5 ± 3.9 kg N ha^{-1} yr^{-1}, respectively) even though corn yields were greater on the fertilizer-based farms. The N mass balance in the corn year was significantly smaller in the diversified farms than in the fertilizer-based farms (55.2 ± 11 kg N ha^{-1} yr^{-1} v. 82.5 ± 5.4 kg N ha^{-1} yr^{-1}; p=0.02).

Corn Grain N

Analysis of corn N content from two years of grain samples collected from farmers' fields during harvest indicated that mean corn N concentrations for the farms in this study did not significantly differ among genetically modified, organic and untreated seed varieties (Table 1.2). In 2009, mean corn grain %N was 1.13 ± 0.02, which was significantly lower than the mean grain %N in 2008 (1.23 ± .02 %N; $p < .0001$). There were significant differences between corn varieties when the %N data was combined with yield data for each field, with genetically modified seed varieties leading to greater harvested N exports from fields (Table 1.2; $p < 0.0001$) than organic or untreated seed varieties due to higher yields.

Legume Biomass, N Content and N Fixation

There was large variation in alfalfa and red clover biomass from samples taken in spring, 2009, just before tillage and incorporation, ranging from 219 – 1003 kg ha^{-1}. Total N in aboveground biomass had a similarly broad range from 9 to 40 kg N ha^{-1}. We entered the sampling data for these individual fields (n=14) into the N balance database, and then estimated the fixed N inputs for remaining fields with spring legume incorporation using the mean values for red clover (33.6 kg N ha^{-1}) and alfalfa (22.4 kg N ha^{-1}) N content, which had been adjusted based on the assumption that 70% of aboveground N was from fixation.

Alfalfa summer biomass over the two sampling years varied from 1.2 to 4.6 Mg ha^{-1} at each hay cutting and red clover biomass ranged from 1.9 to 4.6 Mg ha^{-1}. Alfalfa aboveground biomass was, on average, 3.38 ± 0.15 %N and red clover aboveground biomass had a mean of 3.14 ± 0.11%N. For the simulated hay samples, in which we separated aboveground biomass into

Table 1.2. Observed means and standard errors for corn %N and harvested N (kg ha^{-1}) based on field yields for 2008 and 2009 grain samples of certified organic, genetically modified, and chemically untreated seed varieties. Stars indicate significant differences (*** $p < .0001$) among varieties within a year, and different letters indicate significant differences between years.

	Fall 2008 Corn Harvest			Fall 2009 Corn Harvest			
	Mean	SE	n	Mean	SE	n	Mean (2008-2009)
% N Organic	1.22	0.04	22	1.08	0.04	21	
% N GM	1.25	0.02	31	1.15	0.02	35	
% N Untreated	1.20	0.04	8	1.15	0.02	12	
Mean	1.23 a	0.02	61	1.13 b	0.02	68	1.18
Harvested N Organic (kg ha^{-1})	88.0	6.6	22	80.2	5.1	20	
Harvested N GM (kg ha^{-1})	131.0***	4.5	31	121.9***	3.7	35	
Harvested N Untreated (kg ha^{-1})	89.1	14.5	8	96.4	7.4	12	
Mean	110.0	4.6	61	104.9	3.6	67	107.4

"hay" (i.e., the portion removed from the field by farmers, and the component included in our export calculation) and "stems" that would remain in the field, we found that the harvested portion of alfalfa had a mean %N of 3.42 ± 0.12 and the harvested portion of clover had a mean of 3.44 ± .08 %N. The stems (i.e., the portion of the plant below cutting height) were 1.57 ± 0.08 %N and 2.01 ± 0.09 %N for alfalfa and red clover, respectively. The mean aboveground N harvest index for alfalfa was 90% and for red clover it was 81.5%.

Alfalfa N fixation rates from the second hay cutting of 2009 were variable, ranging from 43 – 104% of aboveground N from fixation. Red clover BNF rates ranged from 74 – 96% of aboveground N from fixation (Figure 1.3). When combined with our biomass measurements, these rates translated into a total mean aboveground N from fixation from 33 – 87 kg N ha^{-1} yr^{-1} for alfalfa, and from 32 – 110 kg N ha^{-1} yr^{-1} for red clover (Figure 1.3). We conducted an analysis to determine how sensitive the balance results were to changes in N fixation rates largely due to this unexpectedly large variability in alfalfa rates (see below).

Mass Balances and Farmer Management Practices

Combining the data from corn grain and legume samples with the interview data and assumptions from the scientific literature, we found large variation in five-year N mass balance across all of the fields included in the database (Figure 1.4). The N mass balance across fields ranged from large surpluses of 149 kg N ha^{-1} yr^{-1}, indicating high potential for N leaching losses, to deficits of 80 kg N ha^{-1} yr^{-1}, which could lead to degradation of soil N pools over time. The median value for the annual N balance of all of the fields was 15.9 kg N ha^{-1} yr^{-1}. This large variability in N balance was expected because the fields spanned the entire continuum of management practices across study sites, and also included the most and least productive fields.

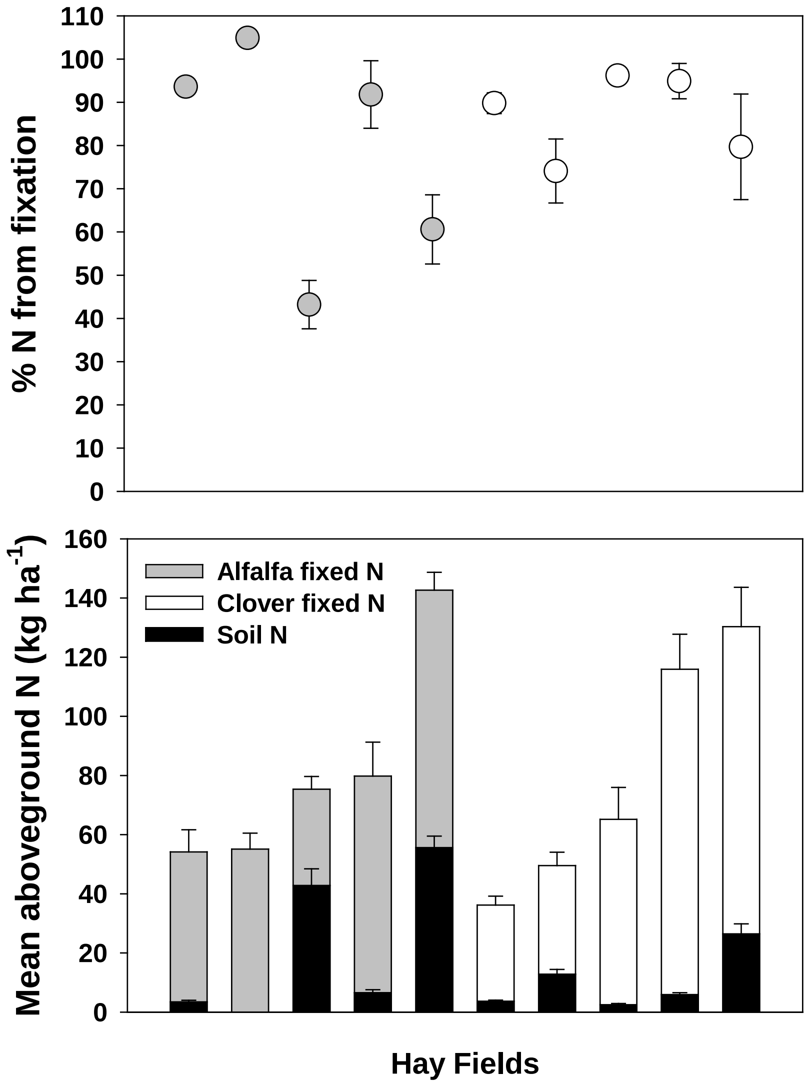

Figure 1.3. Alfalfa and red clover reliance on BNF (% N from fixation) and total aboveground N (kg ha^{-1}) for ten individual hay fields sampled immediately before the second hay cutting in summer, 2009.

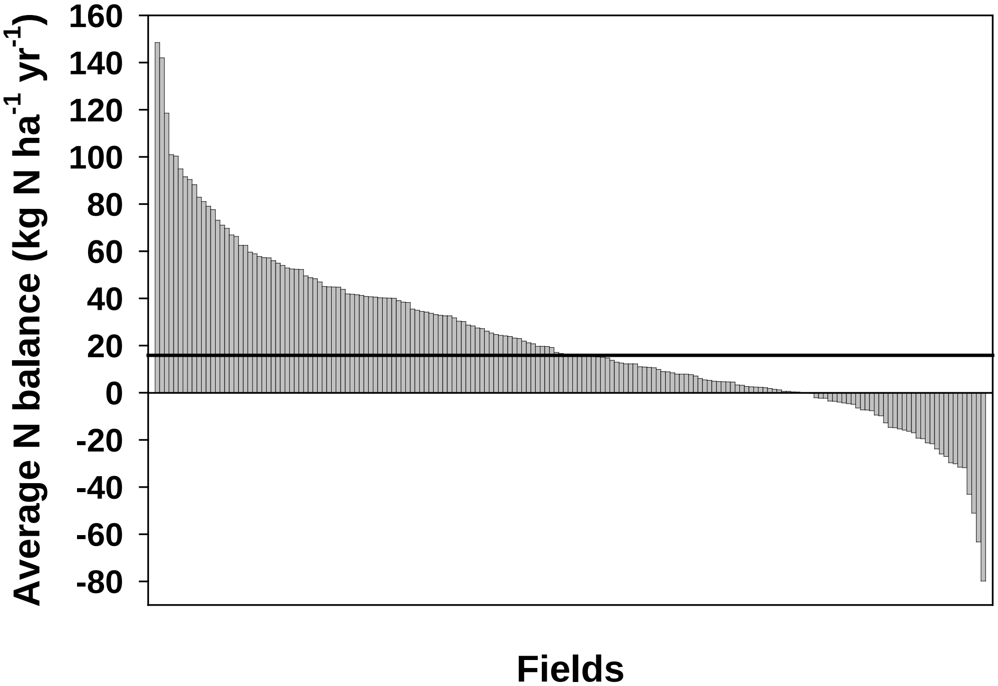

Figure 1.4. Average annual N balance (kg N ha^{-1} yr^{-1}), based on five years of data, for all fields included in the mass balance database, which ranged from surpluses of 149 kg N ha^{-1} yr^{-1} to deficits of 80 kg N ha^{-1} yr^{-1}. The horizontal line indicates the median N balance for all fields, which was 15.9 kg N ha^{-1} yr^{-1}.

The question then, is: what drives this large variability in N balance? And what are the farms that are operating approximately ‘in balance’ doing to achieve small or no N surpluses?

To explain this variability in mass balance we conducted an analysis by N management practice for the different N sources: fertilizer, manure, or legumes (Figure 1.5a). Three common N fertilizer management practices in the MRB—surface broadcast applications of fertilizer, typically UAN; injected applications of anhydrous ammonia fertilizer; and split application and sidedressing of fertilizer, in which fertilizer is applied multiple times, with at least one application occurring post-plant emergence—led to annual N surpluses of 30 to 38 kg N ha^{-1}. The N balances for these three fertilizer-based practices were significantly greater than the mean N balance for the legume-based fields (3.7 kg N ha^{-1} yr^{-1}), supporting our hypothesis that ecological practices which couple C and N biogeochemical cycles have smaller N surpluses. The largest N surpluses were in manure-based fields in which manure was applied in the fall/winter (72 kg N ha^{-1} yr^{-1}); whereas spring application of manure led to N balances (16 kg N ha^{-1} yr^{-}1) that were significantly smaller than the surplus for fall manure and not significantly different from the legume-based fields. The legume-based farms had greater annual N exports over the five-year rotation compared to the other practices, largely due to the dominance of legume hay species in these rotations and large N removals in exported hay.

Nitrogen source and rotation are confounded because fields with a greater proportion of N from legume sources also tend to have more complex crop rotations and greater species diversity. Therefore, we separated source from rotation and Figure 1.5b shows the mass balance analysis with the data categorized by the complexity of the crop rotation, rather than by N source (Figure 1.5a). Rotations with both annual and perennial species had significantly smaller N

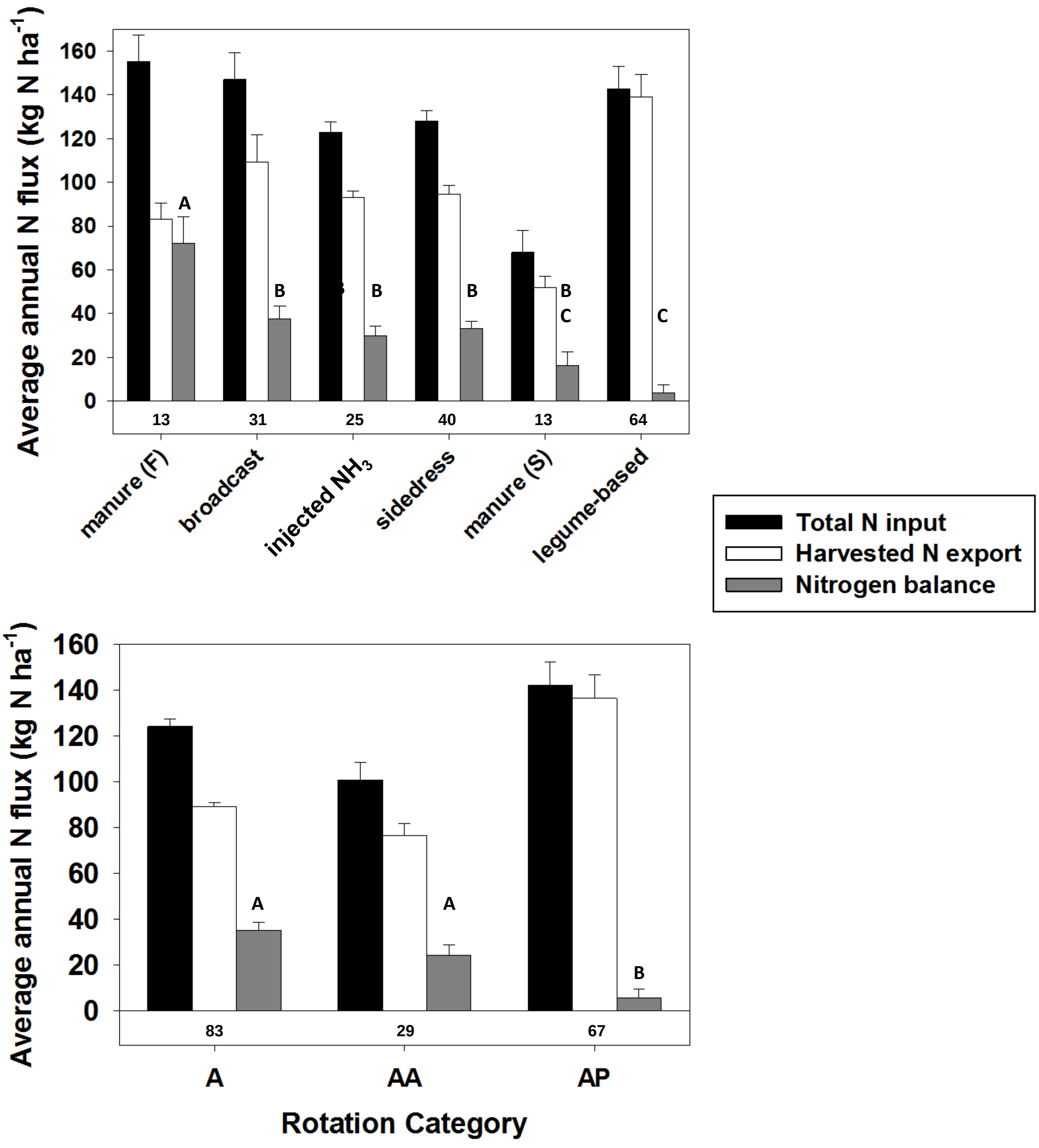

Figure 1.5. Observed means with standard errors for annual N inputs, harvested N exports and N balance (the difference between inputs and exports) based on five years of data for: **a)** different N sources. The x-axis lists different fertilizer, manure and legume management practices, and **b)** different crop rotations: summer annuals only (A); summer annuals and winter annuals (AA); or annuals and perennials (AP). Statistics are for the N balance. The N balances with different letters are significantly different ($p < 0.05$). Numbers listed below the 0 line of the y-axis indicate the number of observations per practice.

surpluses (5.7 kg N ha^{-1} yr^{-1}) than continuous corn or corn-soybean rotations (35.1 kg N ha^{-1} yr^{-1}; $p<0.0001$) and than fields with both summer and winter annuals (24.2 kg N ha^{-1} yr^{-1}).

We also found support for our hypothesis that increasing reliance on legume N sources decreases N surplus and thus potential for N losses. For low annual N inputs, which we defined as N rates of less than 80 kg ha^{-1} yr^{-1}, fields were approximately in balance and there was no relationship between legume N inputs and N balance (Figure 1.6). However, for medium (80-160 kg N ha^{-1} yr^{-1}) to high (greater than 160 kg N ha^{-1} yr^{-1}) N rates there was a significant inverse relationship (R^2 = 0.7 and 0.65, Figure 1.6) in which a larger proportion of N from legume sources led to smaller N surpluses, a trend that highlights the efficiency of legume N sources.

Sensitivity Analysis

Because the most uncertain term in the management-driven N balance calculations is BNF, and because the rates of alfalfa N fixation that we measured in farm fields were particularly variable, we conducted two sensitivity analyses to determine whether the N balance results would change with changes in N fixation rates (Tables 1.3 and 1.4). When we varied alfalfa and red clover N fixation rates to the low and high end of our measured distributions for the fields that fell in the diversified category, the N balance ranged from deficits of 19 kg N ha^{-1} yr^{-1} to surpluses of 24 kg N ha^{-1} yr^{-1}. Thus, even when the two dominant legume species in the database have N fixation rates which are likely unrealistically high, the ecological practices do not fall in the same range of N surplus as the fertilizer-based fields (30-38 kg N ha^{-1} yr^{-1}; mean of 35 kg N ha^{-1} yr^{-1}). Similarly, soybean N fixation rates are known to vary with variety, management, and environmental conditions (Oberson et al. 2007; Schipanski et al. 2010), so we conducted a sensitivity analysis to determine whether adjustments to soybean N fixation would impact the results of the relationship between management and N mass balance (Table 1.4).

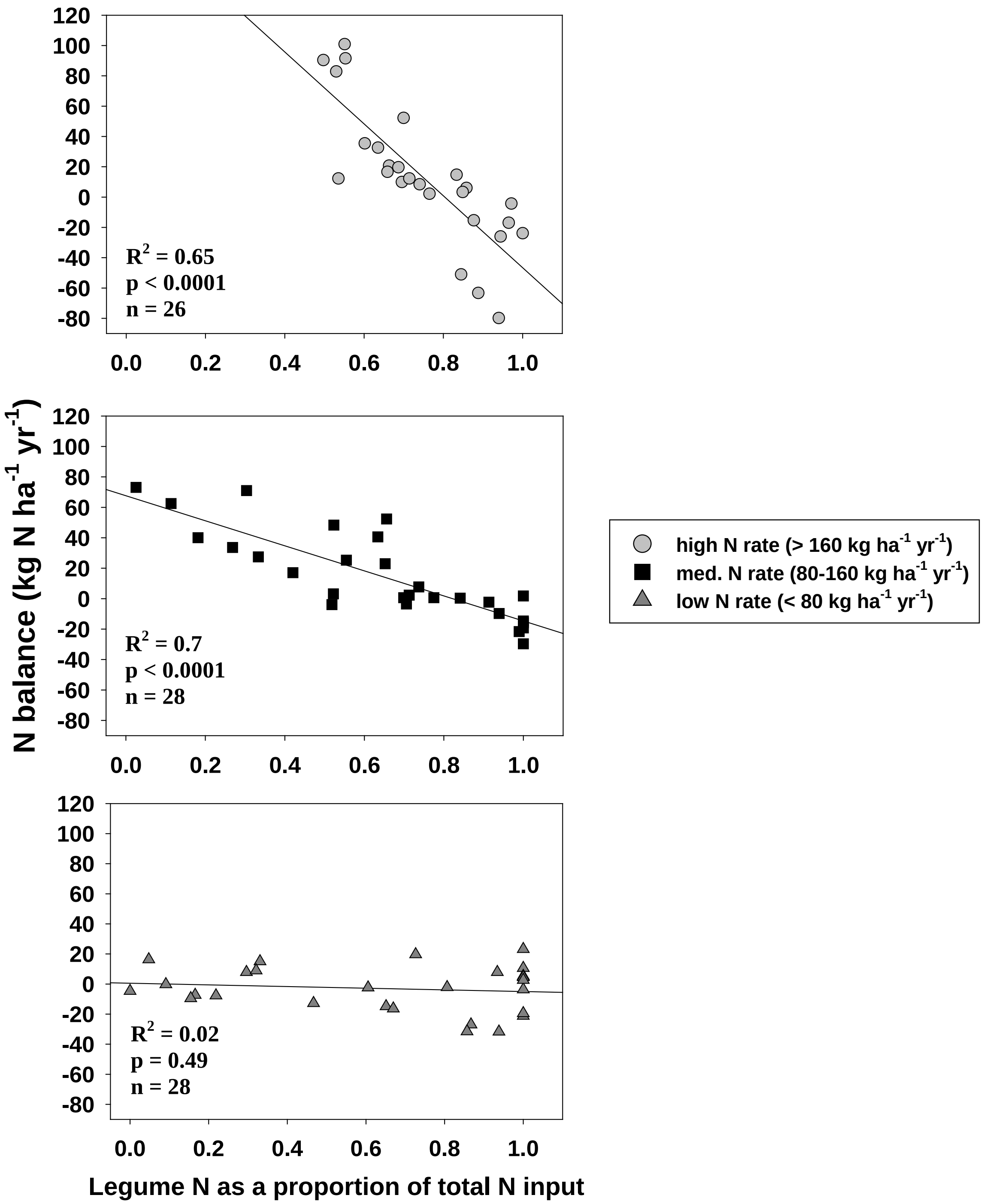

Figure 1.6. The relationship between the proportion of total annual N inputs from legume N fixation (from 0 - 100%) and the N balance (kg N ha^{-1} yr^{-1}) for all diversified and legume-based farm fields, separated by total annual N rate—high, medium, or low.

Table 1.3. Means and standard errors from a sensitivity analysis of N mass balance, for all fields with either legume-based or complex crop rotation practices (i.e., diversified farms), to varying alfalfa and red clover N fixation rates, but holding exports constant.

	Total N Input	**Harvested N Export**	**N Balance**
		kg N ha^{-1} yr^{-1}	
Diversified			
High (75% quartile)	158.8 ± 11.8	135.2 ± 10.0	24.2 ± 4.1
Estimate	140.6 ± 10.0	135.2 ± 10.0	5.4 ± 3.9
Low (25% quartile)	114.9 ± 7.8	135.2 ± 10.0	-19.5 ± 5.1

Table 1.4. Means and standard errors from a sensitivity analysis of N mass balance results to varying soybean N fixation rates. The high value was 80% of total soybean N from fixation, the value used in the final database was 57%, and the low estimate was 50% N from fixation.

	Total N Input	**Harvested N Export** kg N ha^{-1} yr^{-1}	**N Balance**
Fertilizer-Based			
High	127.7 ± 3.0	75.8 ± 1.9	51.9 ± 2.9
Estimate	124.6 ± 3.1	90.1 ± 1.9	34.6 ± 3.2
Low	123.0 ± 3.1	94.2 ± 2.0	28.8 ± 3.2
Diversified			
High	143.0 ± 10.0	132.0 ± 10.4	10.9 ± 4.0
Estimate	140.6 ± 10.0	135.2 ± 10.0	5.4 ± 3.9
Low	141.8 ± 10.0	136.3 ± 10.1	5.4 ± 3.9

Because soybeans occur much more frequently in the industrial, fertilizer-based systems, the impact of changing soybean N fixation from low to high rates was more dramatic on the fertilizer-based farms, changing the N balance from a low of 29 kg N ha^{-1} yr^{-1} to a high of 52 kg N ha^{-1} yr^{-1}. Again, even at the high end of potential soybean N fixation rates for soybean varieties commonly grown in the MRB (i.e., likely no greater than 80% N from fixation; Russelle and Birr, 2004), the diversified fields still have the smallest N surpluses (10.9 kg N ha^{-1} yr^{-1}).

N Inputs from Atmospheric Deposition

In our N balance calculations and analysis we chose to emphasize the largest N fluxes, which are managed by farmers. However, environmental N inputs from atmospheric N deposition (NO_y) are an additional external N input to fields in this region. We therefore also included annual estimates of N deposition inputs for each study site. The mean deposition rate estimates for each site were: 3.8 kg N ha^{-1} yr^{-1} (DML in Iowa); 4.7 kg N ha^{-1} yr^{-1} (DW in Ohio); 2.9 kg N ha^{-1} yr^{-1} (CMN in Minnesota); and 3.7 kg N ha^{-1} yr^{-1} (DA in Wisconsin). The mean deposition input in the Ohio site was significantly larger than the deposition inputs in the Wisconsin and Minnesota sites ($p = 0.0002$). However, adding the N deposition flux to the N balances does not change the results of our analysis of management practices; the N deposition inputs are small, especially relative to the management-driven N inputs, and are essentially similar across sites. Basically, adding this flux increases the N surplus for all fields by approximately 3 - 4 kg N ha^{-1} yr^{-1}.

Response to Environmental Variability: Most Productive and Least Productive Fields

Across study sites, the least productive fields consistently had lower corn yields than the most productive fields, confirming that farmers had, in fact, identified fields with distinct levels of productivity during the interviews (Figure 1.7). The central Minnesota study site had significantly lower corn yields overall compared to the other three sites, likely because our sample captured a drought year, and the Minnesota site was the driest of the four sites. How did farmers respond to this variability on their farms? We expected that low-input farmers with little or no use of commercial N fertilizer would be more likely to respond to environmental heterogeneity with their management practices, either by adjusting total N rates or by adjusting crop rotations. This hypothesis was based on the fact that: i) the diversified farmers tend to be located on more marginal, or less productive soil types (which are more variable) and, ii) diversified farmers rely on ecological interactions in agroecosystems rather than on external inputs that attempt to override ecological processes and existing environmental variability in fields. Our prediction about adjusting rotations was correct; a majority of diversified farmers (66.7%; Figure 1.8A) had different crop rotations on their most and least productive fields, which were tailored to maximize the productive potential of each of those fields given their background environmental characteristics. In contrast, a significantly smaller proportion (31.2%; $p=0.0015$) of the fertilizer-based farmers managed different crop rotations on their most and least productive fields.

We detected no consistent response in terms of total N input rates between least productive and most productive fields in either the diversified or conventional farms (Figure 1.8B). In aggregate, 22.2% of farmers increased N additions, 42% did not change N additions, and 35.8% decreased N inputs on their least productive fields compared to the most productive fields. Though both groups of farmers adjusted their inputs in both directions, a greater proportion of diversified farmers put less N on their least productive field (45.5%), whereas a greater proportion of fertilizer-based farmers put the same amount of N on both fields (47.9%),

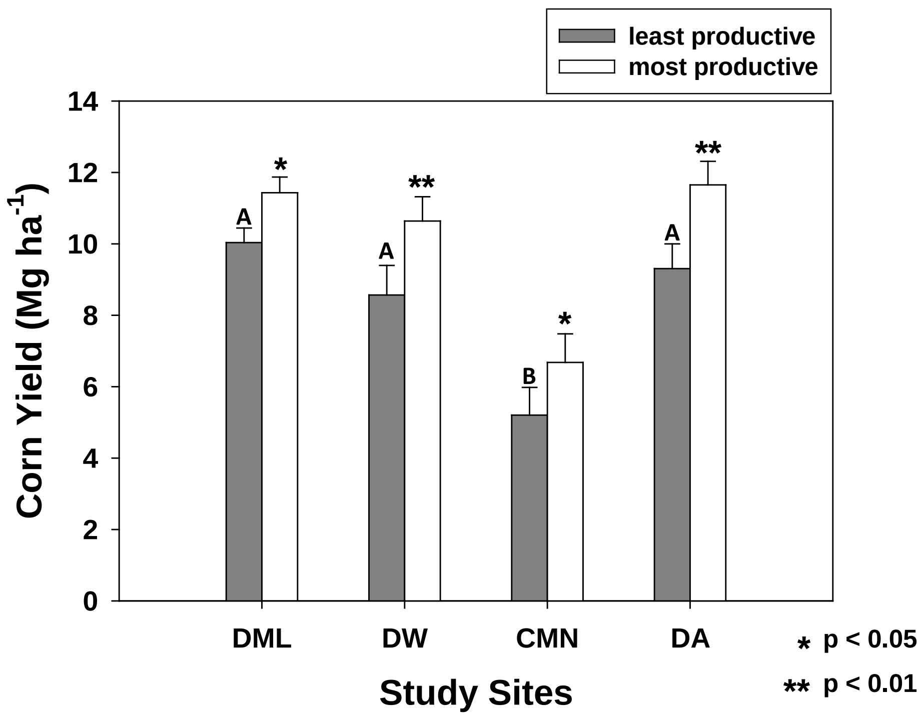

Figure 1.7. Corn grain yields for the most and least productive fields identified by farmers in the interviews. Stars indicate significant differences between most and least productive fields. Different letters indicate significant differences between sites. Yields in the central Minnesota site were significantly lower ($p<0.0001$) than in the other three sites.

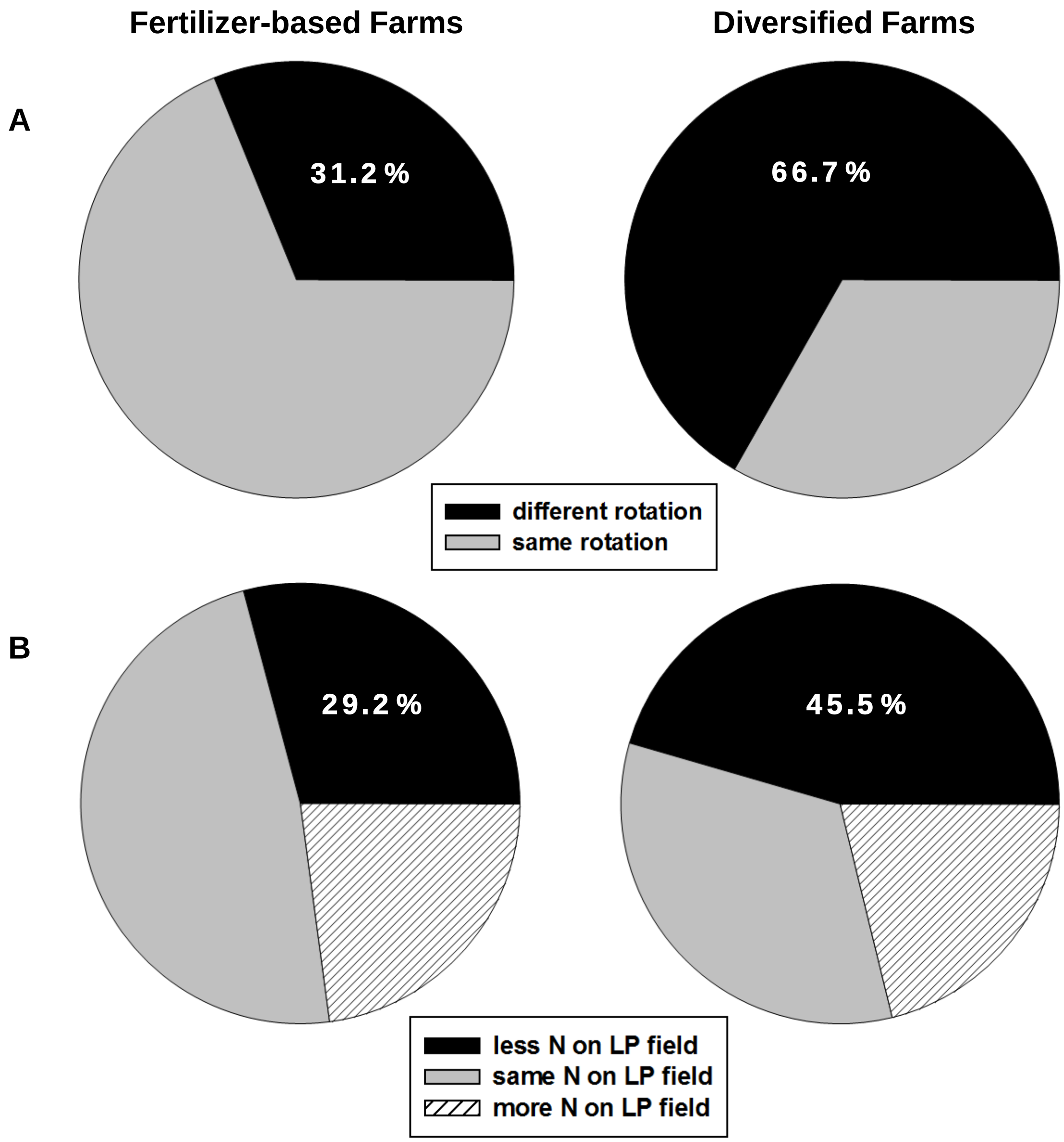

Figure 1.8. Pie charts indicating whether farmers (grouped as fertilizer-based or diversified) responded to environmental variability on their farms by: **a)** managing different crop rotations on their most and least productive fields, or **b)** putting more, less, or the same amount of N on their least productive (LP) field. “Same rotation” means that the most and least productive fields had the same crop rotation.

though these differences in adjustment between fertilizer-based and diversified farms were not statistically significant (p =0.1341).

Since corn yields were consistently lower in the least productive fields across farms, we wanted to determine the consequence of adjusting N input rates on the least productive fields for the N balances (Figure 1.9). The typical cooperative extension recommendation is that farmers should put less N on their less productive fields (i.e., since the yield potential is smaller). We therefore analyzed the difference in N balance (in kg N ha^{-1} yr^{-1}) for the least productive minus the most productive field. A positive number indicates that the N surplus was greater in the least productive field. Overall, farmers who put more N on their least productive field than on their most productive field had significantly greater N surpluses (p=0.02) than farmers who put less N on their least productive field. Therefore, spatial heterogeneity in environmental characteristics at the field scale contributes to the magnitude of N surpluses.

Discussion

Management Practices and N Mass Balance

In this study, we tested whether the trends emerging from meta-analyses and cropping systems experiments would hold in the less-controlled research setting of working grain farms in the MRB. We documented the relationship between a broad gradient of management practices—as actually practiced by farmers in the current sociopolitical and economic context of the region—and surplus N and potential for N pollution. Regional (Howarth et al. 2002), state (David and Gentry, 2000), and county scale (David et al. 2010) mass balances indicate that MRB cropping systems are N saturated (Aber et al. 1989). We therefore know that for the MRB region there is surplus N, but no studies have measured differences in N balance across a large

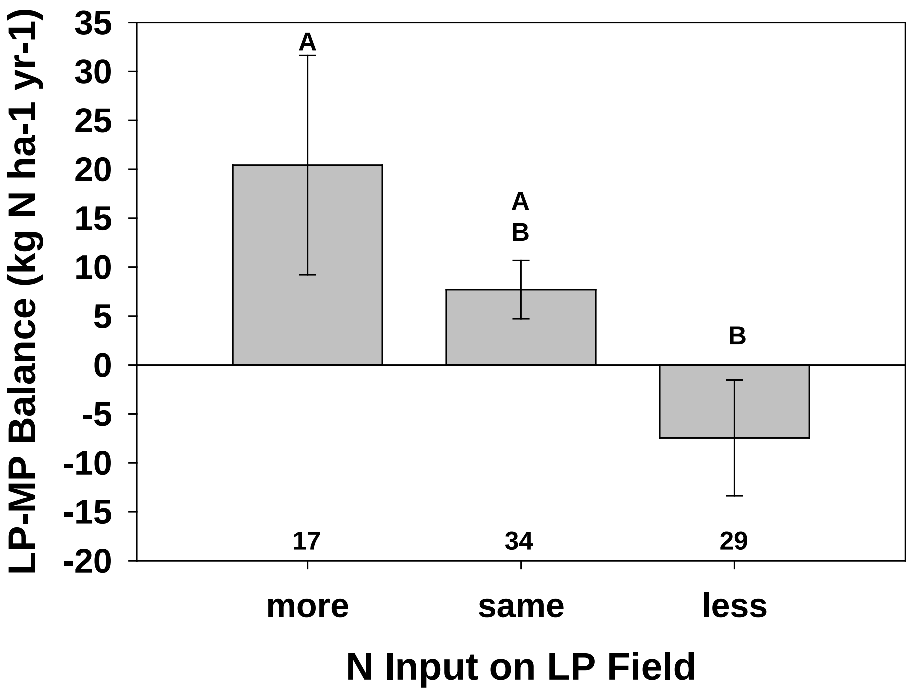

Figure 1.9. Farmers were grouped by whether they put less, more or the same amount of N (total N; kg N ha^{-1} yr^{-1}) on their least productive field. The y-axis is the difference between the N balance in the least productive and most productive fields; a positive number indicates a greater N surplus in the least productive field. Least productive fields with larger N inputs than the most productive field on the same farm had significantly greater N surpluses than fields in which farmers put less N on the least productive field compared to the most productive field.

number of working farms to assess how it varies at smaller scales with management. This type of on-the-ground assessment is important for informing which practices should be promoted to address N pollution from grain agriculture.

Approaches to reducing N losses to surface waters from agriculture in the MRB could be grouped into three categories: removing nitrate after pollution has occurred, adjusting commercial N fertilizer practices, and ecologically-based nutrient management. First, there are practices that attempt to remove NO_3^--N from waterways, through, for example, planting riparian buffers or managing denitrification in constructed wetlands. In contrast, the other two groups of approaches focus on reducing N pollution at the source by improving N use efficiency in farm fields. The dominant approach to reducing N losses from fields in the MRB is via fertilizer-based practices. This includes adjustments to fertilizer management that attempt to maximize crop yield and crop uptake of commercial fertilizer, such as changing the timing, rate, or method of fertilizer application. Finally, though rare on Corn Belt farms, there are ecologically-based nutrient management practices that attempt to balance N budgets by reducing external N inputs, and by adding N together with C, for example, by including annual cover crops in rotation or by applying N in more stable organic forms. In this study, we focused on practices in the latter two categories because our research interest was addressing the root cause of N losses at field and farm scales.

Evidence from quantitative literature reviews of small-scale experiments, and evidence from the few larger-scale cropping systems trials, suggests that ecological practices which re-couple C and N cycles are more effective at reducing N leaching losses. Small-scale controlled experiments, however, do not typically account for a full crop rotation cycle, and cropping systems trials fail to reflect variation in how farmers actually manage their fields. We found wide variability in net N balance across farms in the upper MRB, spanning large N surpluses to large N deficits. Given the differences in management and environmental conditions across working farms, we anticipated this variation. We identified two ecological practices that contributed to reduced potential for N losses: i) reliance on legume N sources and, ii) complex crop rotations that include annual and perennial species. In aggregate, the diversified farms with one or both of these practices had significantly smaller N surpluses than the fertilizer-based farms and farms with fall or winter manure applications. This outcome supports our earlier meta-analysis results (Gardner and Drinkwater, 2009) as well as other literature on the ecological efficiency of perennialization (e.g., Crews, 2004; Randall et al. 1997; Jordan et al. 2008).

Nitrogen source and crop rotation are interconnected, and there was large overlap between fields in the legume-based and complex rotation categories. Legume-based fields frequently had more diverse rotations with perennial species, especially because typical organic farming rotations in the upper Midwest include perennial hay crops (e.g., alfalfa and clover) rather than annual legume cover crops as N sources. Source and rotation are also confounded with total N rate since a primary purpose of external chemical inputs such as N fertilizer is to simplify ecological complexity and maximize the production of a small number of commodity crops. Therefore, having corn, which is a heavy N feeder, fewer years in a crop rotation, and/or increasing the frequency of legumes—which fix their own N—in rotation, means that total N inputs can be reduced. Even though N balance was highly variable across all fields, we found a significant correlation between the proportion of total annual N inputs from legume sources and reduced N mass balance. The diversified farms, therefore, can operate with small N surpluses (e.g., < 10 kg N ha^{-1} yr^{-1}) compared to the mean fertilizer-based surplus of 35 kg N ha^{-1} yr^{-1}.

Together, these results suggest that significantly reducing surplus N in agroecosystems will require reducing N inputs and increasing C availability (i.e., coupling C and N cycles) to support the internal biological mechanisms for storing N, and to improve the efficiency of the internal soil N cycle. More specifically, our results suggest that legumes are a particularly efficient N source (Figures 1.5 and 1.6) because they fix both C and N and thus enhance the ability of these two elements to cycle together like they do in natural ecosystems. In addition to ecological feedback mechanisms such as legume regulation of BNF rates based on soil N availability (Kiers et al. 2003), recent research on the coupling of plant and microbe productivity suggests another ecological reason why legumes might be particularly efficient N sources (Paterson, 2003). In addition to coupling C and N cycles, organic N sources also have distinct temporal dynamics compared to inorganic N sources that improve their N cycling efficiency. Organically bound N is more available than inorganic N in the second year after N addition (Gardner and Drinkwater, 2009). Adding N in more stable, organic forms means that the likelihood of N loss is reduced, especially compared to the pulse inputs of highly mobile N forms in fertilizer-based fields. This suggests that the dominant management emphasis on adjusting the timing and placement of inorganic N inputs has biogeochemical limitations in terms of potential efficiency. Indeed, decades of research focusing on adjusting inorganic N fertilizer applications have failed to reduce the size of the Gulf hypoxic zone (EPA SAB, 2007).

Management-Environment Interactions

Field-level management practices interact with environmental conditions to determine actual potential for N losses as well as the specific loss pathway. We found that differences in N balance across the most productive and least productive fields on a single farm were typically driven by differences in crop yields and N exports, since farmers were not consistently adjusting their N inputs across these two field types (Figure 1.8B). A greater proportion of fertilizer-based farmers managed their environmentally heterogeneous fields with the same practices; however, this trend wasn't significant, and farmers in both categories adjusted N inputs in both directions on their least productive field. We found that farmers who followed the recommendation of reducing N rates on their least productive fields had smaller N surpluses. The diversified farmers were more likely than the fertilizer-based farmers to plant different crop rotations on their most and least productive fields. This is likely because these farmers are more often located on variable and marginal land, and also because fertilizer-based farmers override some of the existing variability on their farms with chemical inputs.

Interpretation of N Surplus in Mass Balances

Based on ecological and biogeochemical theory, we have argued that N surplus is a useful metric for comparing the efficiency of N cycling across diverse farm types. The N saturation literature (e.g., Aber et al. 1989; Fenn et al. 1998) indicates that chronic surplus N additions alter biogeochemical processes in forests, such that when N saturated, they have characteristics that resemble agricultural systems. Fertilizer-based farm fields are N saturated ecosystems because N inputs exceed the ability of the system to either take up or store N internally through plant or microbial uptake (Drinkwater and Snapp, 2007)—for example, on average only about 50% of added fertilizer N is recovered by crops (Smil, 1999) and ^{15}N studies typically find that 38% of added N is unaccounted for at the end of one growing season (Gardner and Drinkwater, 2009). In the Corn Belt, hydrology is a particularly important predictor of N losses due to tile drainage (David et al. 2010). Leaching is an important loss pathway for N (Gast et al. 1978) in this region, varying with soil type, season, and climate.

The other possible fates of surplus N are gaseous losses or storage in SOM, if soil C pools are aggrading. One assumption of N balances that focus on management-driven fluxes is that SOM pools are in steady state, which may not be true. It is likely that fertilizer-intensive corn and corn-soybean farms are not building soil C pools (Neff et al. 2002; Ladha et al. 2011), but other studies have found differences in the storage and composition of soil organic C and N pools between fertilizer-based and diversified farms (Clark et al. 1998; Drinkwater et al. 1998; Mäder et al. 2002; Schipanski and Drinkwater, 2011). The ecological practices tend to couple C and N cycling by adding C and N together (e.g., in organic forms) or by increasing the availability of C-sinks for inorganic N; for example, with practices that build up SOM pools and the capacity of the soil N cycle to supply crop N. If SOM pools were aggrading in our diversified fields, this would accentuate the differences we reported.

In addition to leaching or storage in SOM, the other potential fate of surplus N is gaseous losses. It is difficult to accurately measure fluxes of N_2 and N_2O against the large background N_2 content of the atmosphere, and due to high temporal and spatial variability in emissions (Groffman et al. 2006). Therefore, an accurate understanding of the amount of gaseous N losses from farm fields, and how they are impacted by agricultural management, remains limited (Davidson and Seitzinger, 2006). There is an obvious need to better understand and quantify the partitioning of N losses in agroecosystems, and to constrain these budget terms (David et al. 2009). However, mass balances are a valuable metric for assessing sustainability even when loss pathways are not measured—because losses are problematic whether they are contributing to an air pollution problem or a water pollution problem, or whether wasteful from a farm-level economic perspective.

All of the nutrient management practices explored in this study have the potential for N loss if the cropping system is N saturated (e.g., the N surplus reported here for fall application of manure, which is an organic N source). Long-term use of manure or compost inputs on organic farms as the primary N source can lead to over-application and surplus N additions (Kristensen, 2005; Reider et al. 2000). Improving the synchrony between incorporation of legumes and crop N demand can improve efficiency of legume N sources (Crews and Peoples, 2005). Further, the timing of N surplus is also important. Corn is a vulnerable point in a crop rotation for leaching in both fertilizer and diversified farms (i.e., both had surplus N in the corn year), though the diversified farms may have greater capacity to store that surplus in SOM. Also, the diversified farming systems grow corn less often and have greater plant diversity in time with reduced winter bare fallows, which is the most problematic period for N losses (McCracken et al. 1994).

Mass Balance Approach

Mass balances are a useful metric for comparing management practices, and other studies have calculated field or farm scale N balances on working farms in the U.S. (Barry et al. 1993), or abroad (Van den Bosch et al. 1998), though not for such a wide range of practices for grain farms in a given region. Many of the long-term cropping systems trials also construct N budgets, which have the advantage of measuring changes in soil N storage over long time scales (Paustian et al. 1990; Clark et al. 1998; Drinkwater et al. 1998; Sieling and Kage, 2006; Ross et al. 2008). Our field scale approach, which combined data from farmer interviews, plant sampling and scientific literature, had several strengths. It allowed us to include a large number of farms in the study, but with more rigor than relying solely on literature values for calculations, when we know that some of the N budget terms are extremely variable. For example, our analysis of corn grain N content (Table 1.2) confirms other recent data showing that corn grain N has decreased over the past two decades (David et al. 2010); the standard literature value used to estimate harvested N removal in previously published regional N budgets has been 1.6 %N (David and Gentry, 2000). The decrease in corn grain N content is likely one consequence of plant breeding that has focused on increasing biomass with reduced N rates (David et al. 2010). This change affirms the importance of collecting samples from farms to calculate N balances, because %N in corn grain is a crucial value that has cascading effects on the outcome of the N balances—especially since corn is such a dominant component of most crop rotations in the upper MRB.

The proportion of legume N from the atmosphere (BNF), which represents a new N input to farm fields rather than internal cycling of soil N, was the most challenging and time consuming flux to measure in the N balances. The natural abundance method has been widely applied in recent decades, and is considered a robust tool for measuring BNF rates, despite the complex N cycling processes and environmental and management conditions that affect isotopic signatures in soils and plants (Shearer and Kohl, 1986; Carlsson and Huss-Danell, 2003; Unkovich et al. 2008). Primary sources of variability in the application of the method include selection of reference species and the B values used (e.g., Högberg, 1997, Boddey et al. 2000). We measured B values specific to the legume species and varieties that we sampled, but there are still numerous problems with applying B values from pot experiments to field measurements of BNF including differences in BNF efficiency among *Rhizobium* strains; for instance, we inoculated with one strain, though many different background populations are likely present in the field (Riffkin et al. 1999).

There is clearly a need for improving estimates of BNF inputs to farm fields, for understanding legume-environment interactions, and characterizing legume functional traits to optimize management of BNF on farms (Drinkwater and Snapp, 2007). In a study of this scope, however, it was not possible to explain variability in BNF across management practices and environmental conditions. Instead, we made strategic use of a nested sampling design to quantify this flux using isotope methods in a small number of fields for the two most important legume species in the MRB crop rotations, in order to improve our estimation of this flux and the rigor of the N balances. Further, despite the usefulness of aboveground measurements, uncertainty of belowground fixed N inputs from legume species remains high. Measuring biomass, N content, and BNF rates in roots is challenging and labor intensive, and there are few published studies quantifying this pool (Huss-Danell et al. 2007). In particular, estimating root N inputs from perennial species is complicated by turnover of fine roots, rhizodeposition and exudation. Since alfalfa and red clover were grown for multiple years in most of the diversified rotations, there is likely greater variability in the ages of roots present in the soil and associated variability in decomposition processes and the timing of nutrient release. Further, in perennial species, nutrients are retranslocated within plants before tissue senescence each season, which contributes to enhancing nutrient cycling efficiency (Crews, 2004), but also makes it difficult to understand soil nutrient cycling dynamics in perennial ecosystems. We made a best estimate of belowground inputs from exudation, rhizodeposition, and root death, based on other published studies, because this is more accurate than ignoring this large N source. However, belowground legume N inputs likely represent the greatest source of uncertainty in the balances. The sensitivity analysis demonstrating that our primary conclusion about the relationship between practices and net N balance doesn't change even at very high BNF rates gives us more confidence in our findings despite the uncertainty of this balance term.

Implications for Agricultural Research and Policy

Mass balances focused on the largest N flows controlled by farmers have potential applications for agricultural policy. For instance, one approach could be to use N mass balances as a simple verification tool to determine farmer eligibility for incentive programs designed to conserve nutrients and preserve water quality. United States farm policy has followed a practice-based strategy that targets single practices rather than promoting farming systems that employ a suite of synergistic practices. In contrast, the work reported here offers a more holistic, ecosystem-based framework to farm policymakers. Because surplus N is an indicator of potential for N loss, ecological nutrient management involves balancing N inputs and exports as much as possible (Drinkwater and Snapp, 2007). Instead of taking this ecosystem approach to addressing N losses in the MRB, several decades of research have focused predominately on managing applications of inorganic N fertilizer. And agricultural policy in the U.S. has focused on increasing the production of a small number of grain commodities, an emphasis that has had large ecological costs (Matson et al. 1997, MEA, 2005). For instance, the spatial distribution of federal crop commodity payments is correlated with the distribution of N leakiness from counties in the MRB (Booth and Campbell, 2007; Broussard and Turner, 2009; David et al. 2010). These linkages underpin and reinforce the current trajectory of agriculture in the region, and this unsustainability has led to important questions about ecological efficiency and interest in alternatives from a growing number of stakeholders.

Our findings contribute to a growing body of evidence that ecological approaches to nutrient management, which re-couple C and N cycling, have the greatest potential to address the N pollution problem in the upper MRB. We found significantly lower corn yields in the diversified farms compared to the fertilizer-based farms; however, total harvested N exports over the five-year crop rotation were not different indicating that the diversified systems are also highly productive. There are a number of possible reasons for the lower corn yields in the diversified fields including: i) breeding and selection of corn varieties under high input conditions has favored varieties that are most productive with chemical inputs and that require surplus N additions to meet yield goals (David et al. 2010); ii) in addition to the lack of plant breeding targeting alternative agroecosystems, there has been a lack of research exploring options for optimizing nutrient management in organic and diversified cropping systems, which rely on nutrient sources that must mineralize before they are plant-available (Drinkwater and Snapp, 2007); iii) for farms that had recently transitioned (or were transitioning) from a fertilizer-based management history, it takes years for SOM pools that support crop production under legume-based management to build up and reach steady state (Drinkwater and Snapp, 2007). Several recent literature reviews indicate that ecological management practices have the potential to simultaneously improve environmental outcomes and maintain crop yields (Tonitto et al. 2006; Badgley et al. 2007).

Further, it is now well-accepted that a singular focus on increasing crop yields not only leads to environmental costs (MEA, 2005), but is also insufficient for meeting human food needs. Under current conditions, poverty is the root cause of hunger, where the distribution of and access to food is the most critical factor (Smil, 2000; Badgley et al. 2007). In the upper Midwest, less than 25% of crop production goes directly to food production (i.e., food directly consumed by people) compared to uses such as animal feed, fiber, or bioenergy crops (Foley et al. 2011). Allocating grains from highly productive cropland to animal feed is ecologically inefficient and actually reduces the world's potential food supply (Foley et al. 2011). A shift in focus is therefore needed to design and implement policies that support *food* production while meeting multiple environmental and social goals. In addition, conservation policies could also be designed to harness the co-benefits of ecological practices such as use of legume N sources and complex crop rotations with annual and perennial species. For instance, these practices likely correspond to increases in soil C stocks, which could have additional benefits for mitigating climate change, though there is a need to more accurately measure changes in soil C pools. Further, legume N fixation is solar powered, whereas commercial N fertilizer manufacture and use requires a large input of fossil fuel energy—accounting for approximately 50% of the total CO_2 emissions from industrial agriculture (West and Marland, 2002).

Despite the evidence from small scale and short-term agronomic experiments, from long-term cropping systems trials, and evidence from the on-farm work we reported here, ecological nutrient management practices remain extremely rare on farms in the MRB. Nitrogen pollution is therefore not simply a technical problem (Allen, 2004). There is already more knowledge about the ecology of cropping systems than what is being applied to agricultural landscapes; the management practices that are the most efficient are not supported by current policies, and are uncommon. Clearly the social and political context cannot be separated from the ecological sphere. This is true of most agri-environmental problems, not just nutrient pollution. Farms are nested within ecological and social relations of production, and those are nested within the history and policies that have shaped the industrialization of agriculture in the MRB. Thus multiple disciplinary perspectives are needed to understand environmental problems like Gulf hypoxia, as well as involvement of diverse stakeholders and actors outside of academia.

Conclusion

Nitrogen is an important nutrient limiting crop productivity that is widely applied to industrial agricultural landscapes. Grain farms in the upper MRB contribute the majority of the N load to the Gulf of Mexico, causing eutrophication. Despite decades of research and efforts to promote best management practices for commercial N fertilizer, N leaching losses have not been reduced and the hypoxic zone in the Gulf of Mexico has persisted. Our ecological, on-farm research contributes new information on the relationship between field-level management practices and N mass balance in the basin. We found that management practices that couple inputs and cycling of C and N, such as use of legume N sources and complex crop rotations that include perennial species, more effectively reduce the potential for N pollution from farm fields than adjustments to N fertilizer management in corn-soybean rotations. This on-farm evidence—from sites in the area of the MRB known to contribute disproportionately to the hypoxia problem—corresponds with the results of small-scale studies (e.g., the ^{15}N literature) as well as field-scale trials at agricultural experiment stations, suggesting that policies should promote management practices that couple C and N cycling in farm fields. On-farm, and systems-based research is more complex and time consuming, but the benefit is that the results are situated within a real-world scenario that, by definition, integrates human dimensions of management with ecological processes.

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CHAPTER 2

INTERSTITIAL INNOVATION IN THE MISSISSIPPI RIVER BASIN: EXPLORING HOW FARMERS TRANSITION TO ECOLOGICAL MANAGEMENT

Abstract

The industrialization and specialization of agriculture in the Mississippi River Basin (MRB) has led to widespread environmental and social costs. As a prime location for human interaction with the environment, agrifood systems represent a clear hybrid of social and natural processes. Through an integrated socioecological lens, this study explored how a subset of alternative grain farmers and rotational graziers in two contrasting regions of Iowa (central and northeastern) transitioned to management practices with the greatest promise for providing ecological benefits in the region. By applying a resource-based perspective to analyze qualitative interviews conducted with farmers between 2008 and 2010, I identified the internal (i.e., micro-level) and external resources and strategies that farmers harnessed to develop opportunities for, and overcome barriers to, transitioning to alternative practices within the macro political economic context of the neoliberalization of agriculture in the MRB. To enhance resilience and mitigate risk, alternative farmers in Iowa cultivated farm-level biodiversity and enterprise diversity. They developed new cognitive and psychological competencies such as ecological thinking and new ways of relating with others. Finally, they overcame barriers to innovation by developing external network linkages with peers, knowledge organizations, and federal policy. I argue that these approaches constitute innovation within the interstices of the dominant economic logic of agriculture in the region, creating new spaces for transformation toward sustainable agrifood systems that provide a greater range of ecological and social benefits. Compared to central Iowa, these alternative agricultural practices have become less interstitial in the northeastern region,

where a geographic cluster of alternative farms and supportive institutions facilitates more efficient transitions to alternative modes of production.

Introduction

Agriculture is at the center of increasing public interest with the confluence of global climate change, food and energy crises, and the co-occurrence of hunger and obesity. The conversation about how to address these crises is contentious. Dominant neoliberal policies advance industrial food production, though it is clear that input-intensive approaches to agriculture contribute to environmental degradation (MEA, 2005). There is now greater understanding of the potential for agroecological management to improve the sustainability of food production (IAASTD, 2008), and a growing number of farmers, citizens, academics, and other actors promote the benefits of more socially just models of agriculture based on ecological principles (e.g., IAASTD, 2008). In addition, agrifood system issues are perhaps uniquely positioned to catalyze social movements and social change since food is such a personal commodity, which we interact with so frequently and which is incorporated into our bodies (Allen, 2004). Problems and solutions within agriculture are both social *and* natural, and require an integrated perspective and a pluralistic epistemological frame to be fully understood.

In this paper I explore how farmers in the Mississippi River Basin (**MRB**) are transitioning to emerging alternative production systems that have the greatest promise for providing ecological benefits. Water pollution is one of many well-documented environmental consequences of industrial agriculture, and the intensive grain farms of the Corn Belt states in the upper MRB are the primary source of nitrogen leaching losses to the Gulf of Mexico, which cause a hypoxic (or, low oxygen) zone to form each summer (David et al. 2010). This problem is socioecological. The systemic intensification of agriculture in this region has been shaped by the historical development of commodity programs in the U.S., incentives provided by the expansion of a globalized market for agricultural commodities, and by a lack of regulation of non-point source pollution through, for instance, Clean Air Act and Clean Water Act legislation, and the resulting externalization of pollution and associated costs. Ecologically, this has meant replacing plant species diversity, ecological management, and reliance on internal ecological processes with external chemical inputs, such as nitrogen fertilizer. The low diversity of these agricultural systems—which also have long winter bare fallow periods—combined with pulse inputs of soluble inorganic nitrogen fertilizer, leads to their 'leakiness' (Drinkwater and Snapp, 2007).

Iowa is one of several major commodity-producing states in the Corn Belt of the upper MRB, which is the region that contributes the bulk of the nitrogen load to the Gulf of Mexico (David et al. 2010). The most intensively farmed regions of Iowa are the former wetland, or "prairie pothole" soils, which have very high levels of organic matter and are extremely productive when drained. Over the past two decades, however, Iowa has also become a hotspot of innovation around alternative agrifood systems (Hinrichs, 2003) compared to neighboring Corn Belt states. Research describing this innovation frequently centers on the growing local foods movement, which has created new markets such as community supported agriculture (CSA) schemes and farmers markets (e.g., Hinrichs, 2003). However, annual cereals and legumes are the dominant crops in the MRB, and are also the primary source of pollution in the region (Turner and Rabalais, 2003). Grains are planted on approximately 70% of cropland worldwide, and comprise a similar proportion of human calories (Glover et al. 2010). There is no question that developing ways to produce grains more sustainably presents a significant and urgent challenge. Therefore, successful alternative grain farmers in the MRB are potentially "small causes" that could lead to large effects if they are able to inform and contribute to transforming the grain cropping systems of the region.

Industrial grain farms of the Midwest are part of a larger grain-livestock commodity complex that funnels a large proportion of food calories through animals raised in confinement. Transforming grain cropping systems to reduce pollution will thus require similarly radical change within animal production and most likely a corresponding shift to diets lower in animal protein (Pimentel, 2004). One largely farmer-developed alternative to animal production in feedlots is management intensive rotational grazing (MIRG) of animals, in which pasture is the primary animal food source during the growing season, or longer (for example, through 'stockpiling' hay or pasture). The key elements of MIRG are short grazing periods with high animal stocking densities and adequate time for plant recovery between grazing periods (Murphy, 1995). Holistic Management International[2] is an example of a prominent farmer organization that integrates grazing knowledge from the work of early pioneers such as French scientist Andre Voisin (Voisin, 1959) with on-farm experimentation and more recent ecological and management knowledge (e.g., Savory, 2008). Grazing animals on perennial pastures remains understudied, but has many potential benefits, especially because perennial ecosystems are known to provide a wider range of ecosystem services and functions than annual cropping systems (e.g., Crews, 2004). In this study I focus on alternative grain cropping and rotational grazing systems in Iowa. Even though animals and grain crops are currently physically separated on the landscape, they are in fact linked, and sustainable agriculture will require their re-integration (Russelle et al. 2007).

[2] http://www.holisticmanagement.org/

The central objective of this work is to analyze how a subset of grain farmers and graziers in Iowa transitioned to ecological management practices with the greatest potential to reduce nitrogen pollution of waterways. I explore this 'bottom up' change in two contrasting regions of the state—the flat, prairie soil region of central Iowa, and the steeper Driftless Area in northeastern Iowa. These farms represent a consequential countertrend to the dominant context of industrial agriculture, yet the transformative potential of these pockets of innovation is an open question; that is, do they hold promise for catalyzing more widespread resistance, or are they simply isolated—on the margins within a highly industrialized landscape? Because the management practices I focus on in this study are currently so rare in the upper Midwest, some might argue that devoting serious scholarly attention to them is unwarranted or impractical. However, in this paper I argue that a detailed understanding of these emerging alternatives is important precisely because of the need to create more political and academic space for taking these approaches seriously—ecologically, they have the greatest potential to address current global crises. Though they are marginalized, we can locate power among these alternative farmers and their networks, and understand the ways in which they are contributing to interstitial and bottom up change in this important agricultural region. By making a social decision to support multiple ecosystem services on their farms, their effects might therefore be disproportionate to their size (Allen, 2004; Wright, 2010).

Farmer Transitions

Iowa exemplifies the dominant trends of industrial agriculture. The total number of farms in the state has declined dramatically—from more than 200,000 in 1950 to fewer than 100,000 by the late 1990s—and the number of alternative farmers in the state is extremely small. Since organic agriculture is a legally-defined category, more data is available on organic production than data on ecological or alternative farming systems. In Iowa, 13.8 million acres of corn and 8.6 million acres of soybeans were harvested in 2007, and in 2008 the total acreage in certified organic grains and beans together was just under 63,000. Certified organic livestock in Iowa make up about 4% of total organic livestock in the U.S. (USDA, 2008). On a national scale, organic agriculture has dramatically increased since the beginning of the 1990s, from under one million acres of certified organic land to nearly five million acres (2.7 million acres of cropland and 2.1 million acres of rangeland and pasture) in 2008. The adoption rate of organic agriculture during the past decade has therefore been high; however, the total amount of organic land remains low, representing 0.57% of total U.S. cropland (USDA, 2008). For grain crops the proportion is even smaller. In 2008, the acreage of certified organic corn comprised 0.21% of all land in corn in the U.S. And, together, grains, beans and oilseeds make up about one million of the total certified organic acres nationally. In the same year, certified organic livestock (not including poultry) made up about 0.28% of the total production (USDA, 2008).

A large literature has addressed the question of *why* farmers convert to alternative agriculture, especially organic agriculture (e.g., Fairweather, 1999; Darnhofer et al. 2005; Padel, 2008). These studies seek to understand the rationales underpinning farmers' choices of farming methods. They argue that supporting the growth of alternative agriculture will require understanding why farmers do or do not adopt these practices. Many farmers are motivated to transition to alternatives for economic reasons (Greene and Kremen, 2003), health reasons (Lockeretz and Madden, 1987), or for ideological reasons. For example, Glenna and Jussaume (2007) described different value orientations held by organic and conventional farmers in Washington State. In another study conducted in Iowa, Bell (2004) interviewed farmers who were members of the organization Practical Farmers of Iowa. He found that the majority of farmers who converted to alternative agriculture were motivated by an economic or health crisis. Yet Bell also argued that the large number of surveys conducted in recent years aiming to predict farm-level conversions “haven’t found much” (p. 159). For example, even though economic motivations are an important commonality among farmers who transition, *most* Iowa farmers experience economic problems and do not shift to alternative agriculture. That is, because alternative farmers were more *similar* to conventional growers than they were different before the transition (in terms of demographic factors and exposure to the stresses and risks of farming) it is still an open question why these factors lead to conversion for some and not for others.

In a review of the organic conversion literature, Padel (2001) notes a temporal shift in farmers’ reasons for transitioning to organic production. In earlier studies, farmers who transitioned emphasized religious and philosophical motivations, whereas recent research has found that farmers who transition tend to have larger farms and cite economic reasons for transitioning to organic agriculture (Padel, 2001). Therefore, while pioneering organic farmers may hold strong value commitments, those who transition later may be more motivated by opportunities to increase farm profitability, potentially reflecting a trend toward appropriation and conventionalization due to the entry of agribusiness into the organic sector (Guthman, 2004). Economic incentives for specific ‘alternatives,’ such as certified organic grain production, are indeed growing. However, external incentives such as price premiums do not translate directly into action on the ground (Wolf and Primmer, 2006). Given the breadth of existing industrial and alternative management practices on farms, and how they articulate with various socioeconomic and political factors, it is not surprising that the reasons why farmers transition to new farming systems are diverse and reflect a continuum of commitment to ideals or values (Darnhofer et al.

2005). At the level of individual farmers the motivations and competencies for transitioning to agroecological alternatives are shaped by a complex suite of psychological and farm-level factors that interact with external opportunities and risks. It is important, therefore, to move beyond simple individual-level explanations of transition, which neglect a wide range of socioeconomic, political and ecological processes.

Transition Processes as Collective, Relational, and Site-Specific

To more fully explore farm-level transition dynamics, I argue it is useful to shift our gaze towards *how* in addition to *why* these transitions occur. This systems perspective allows greater opportunities to locate the process of farm-level decision-making in a macro historical, political economic and cultural context. Social theories and methods that prioritize participatory research approaches, social learning and socially- and historically-situated analyses are all important modifications changing the direction of research on and analysis of agricultural innovation. For example, as Padel (2001) notes, organic farming is better understood as a bottom up and farmer-driven movement that represents a complex, systemic change rather than the adoption of single technologies. It is therefore necessary to draw upon more recent relational conceptualizations of technical change that are better equipped to explain innovation towards agroecological alternatives in a more holistic and situated way.

In this study I apply and extend the resource-based conceptual model of innovation that Wolf and Primmer (2006) developed to describe mechanisms leading to change and action in the Finnish forestry sector. Development of new material practices that contribute to biodiversity conservation and ecological sustainability requires the creation of new knowledge and skills (i.e., know-how). Wolf and Primmer elucidated a suite of competencies, or resources, which mediated learning and the process of transitioning to management practices that conserve biodiversity for forest management actors. They argued that transitioning to more sustainable management practices is a collective, or social, process that depends upon the acquisition of resources, which improve the efficiency of learning and the formation of competencies. In their Finnish empirical work, they identified three general categories of competencies, which were grouped as either internal or external to the organization. For example, human capital and organizational routines are internal, whereas network linkages represent external competencies, or the ability to harness outside resources that mitigate risk and enable change. The key point is that resources mediate the formation of competencies, which enable actors to promote and enact new, more sustainable land management practices.

Similarly, resources and competencies that are internal to farms interact with relationships between farmers and external organizations to support transitions to alternative forms of production. Simple explanations of the transition process, such as economic incentives or individual-level psychological motivations, are not sufficient to understand and explain the complex dynamics of innovation. In this study, I adapted and applied the conceptual model of competencies in Wolf and Primmer (2006) to explore how farm-level transitions to agroecological practices occur in the MRB. I broadly conceptualized the internal competencies on farms as including not only material and ecological resources (e.g., investments in new equipment or training), but also epistemological and psychological factors such as values, motivation, discourses, and self-understandings. Primmer and Karpinnen (2010) expanded the resource-based model described above to include the range of values and beliefs that professional foresters hold about biodiversity conservation, as well as their social context, to understand why some foresters voluntarily exceed the minimum requirements for conservation.

However, while these authors took a quantitative approach to assessing organizational resources, I used qualitative methods to gain an in-depth understanding of the internal competencies which are not easily measured or counted. Individual motivations and farm-level resources also intersect with a wide range of actors, such as private companies, Land Grant Universities, agri-environmental organizations, service providers, and many others. Farmers thus derive resources from numerous public and private actors, and it is important to understand how relationships between farmers and external organizations enable or constrain innovation.

In addition to emphasizing collective action to move beyond individual-level explanations of transition, it is also analytically helpful to challenge the society-nature dualism that characterizes modernist ontology (Goodman, 1999). Rather than viewing humans and nature as separate domains, I view particular arrangements of humans and non-humans as networks (Goodman, 2001; Castree, 2005). This study is grounded in ecological theory and interconnected with a related ecological study (see Methodology section). In addition, this analysis of farm-level transitions explores the role of the biophysical environment as both a constraint and opportunity for agroecological innovation in two ecologically contrasting regions of Iowa. Thus agency is "a property that emerges through interactions of people and objects and through relational networks" (Steins, 2002, p. 413) rather than being a particular (e.g., social or natural) characteristic of any one component (Murdoch et al. 2000). Other studies have engaged with this concept of "ecological agency" in the development of alternative food systems (Wittman, 2009) or in sustaining conventional agrifood systems (Galt, 2010). Giving symmetrical analytical attention to the materiality of nature is central to understanding agricultural innovation.

Further, alternative agrifood networks, and their biophysical and institutional environments, are spatially embedded, with site-specific opportunities and barriers for transitioning in different locations within Iowa and the MRB more broadly. Other studies have demonstrated the importance of clustering to explain growth of alternatives in areas with concentrations of resources, organizations or policies that support these types of activities (Porter 2000; Martin and Sunley, 2003; Centonze, 2010). Social processes and institutions co-evolve with particular biophysical characteristics to form such clusters of innovation, shaping where and how alternatives emerge or succeed. Given this real-world complexity, it is necessary to explore the role of socioecological factors at multiple levels of organization to explain the adoption of complex alternative farming systems. In addition, it is useful to compare contrasting regions to understand how the opportunities and barriers to transitioning to alternative production vary with differing arrangements of socioecological and political resources.

Political Economic Context

Individual farms and the agrifood systems of the MRB are nested within a macro-level political economic context favoring the globalization and specialization of agriculture. The neoliberal, or corporate, food regime (McMichael, 2005) of the contemporary moment hinges on a tension—clearly present in this region—between the dominant trend towards specialized production of homogeneous commodities for global markets and the expanding interstitial presence of ecological, diversified production for local and regional consumption, though a broad range of agricultural practices falls within each of these categories. In the upper Midwest, the balance of power among actors at these two extremes is highly asymmetrical, and it therefore makes sense that only a small number of farmers would transition to alternatives. Choosing to manage farms differently is risky and challenging in this region and the pressure against doing something different is large given the socio-technical regimes in which farmers are situated and the agglomeration of power that has constructed the landscape. Elite actors, including transnational agribusinesses and commodity organizations, currently control the majority of agricultural production in the upper Midwest and the discourse about how to manage agroecosystems. A primary place of influence for these powerful interest groups is controlling agricultural markets with unprecedented levels of concentration, as well as dominating the U.S. Farm Bill negotiations. The result is a federal farm policy that directs taxpayer dollars to surplus commodity production, and which is unsupportive of the practices that are the focus of this study.

Methodology

This project was interconnected with an ecological study focused on nitrogen cycling and nutrient management in agricultural ecosystems, which, in turn, was nested within a larger interdisciplinary collaboration centered on the problem of hypoxia in the Gulf of Mexico. Beginning in 2007, and continuing through 2010, I conducted agroecological research relating management to the potential for nitrogen pollution from farm fields with grain and grazing farmers spanning the full range of practices found in the upper Midwest. The ecological interviews (125 total) were conducted in study sites in Iowa, Minnesota, Ohio, and Wisconsin and were supplemented with analyses of plant samples to calculate field-level nitrogen budgets. The goal of the qualitative work presented here was to investigate in greater depth the alternative farming systems that had the most efficient nitrogen cycling and smallest potential for nitrogen pollution based on the ecological findings (Blesh and Drinkwater, *in prep*.). Farmers selected for this qualitative study can be considered outliers in the Corn Belt because they had successfully transitioned to the most ecologically efficient alternative practices in the typology (Figure 2.1): i) management intensive rotational grazing, which is an agroecosystem based on perennial species, or ii) certified organic grain farms with complex crop rotations including annual and perennial crops, and greater than fifty percent of total nitrogen inputs from legume sources. The sample is therefore comprised of a highly specific subset of practices with the greatest potential to address nitrogen leaching losses from agriculture in the upper MRB (Blesh and Drinkwater, *in prep.*); these practices are also currently extremely rare on agricultural landscapes in this region. Agricultural practice has clearly not incorporated the full extent of ecological knowledge that is available to address the problem of nitrogen pollution, and this study is just one of many possible examples highlighting that the barriers to sustainable food systems are as much social as they are technical (Allen, 2004).

The work presented here extends and complements the natural science research, to more fully explore *socio*ecological dynamics on the most efficient alternative farming systems in the Corn Belt. Though rarely acknowledged by natural scientists, *all* theories have different ethics and values; I integrated critical social science perspectives, which recognize the normative dimension of the research process, into my work. Through this research, I aimed to practice an engaged agroecology that seeks to understand the sociopolitical relations of food systems and to locate power, and potential leverage points for change, in those relations (Vandermeer and Perfecto, 1995; Rocheleau, 2008; Perfecto et al. 2009). In addition, the ecological theory and data used to select farmer-participants for this qualitative study brings rigor and a nuanced perspective that is often lacking in the coarse categorizations of the sociology of food systems (e.g., “conventional” or “organic”). For instance, as Guthman (2004) points out, “the very existence of agribusiness participation in the [organic] sector points to the fact [that] deeper meanings of organic farming are not codified in existing rules and regulations” (p. 308). By

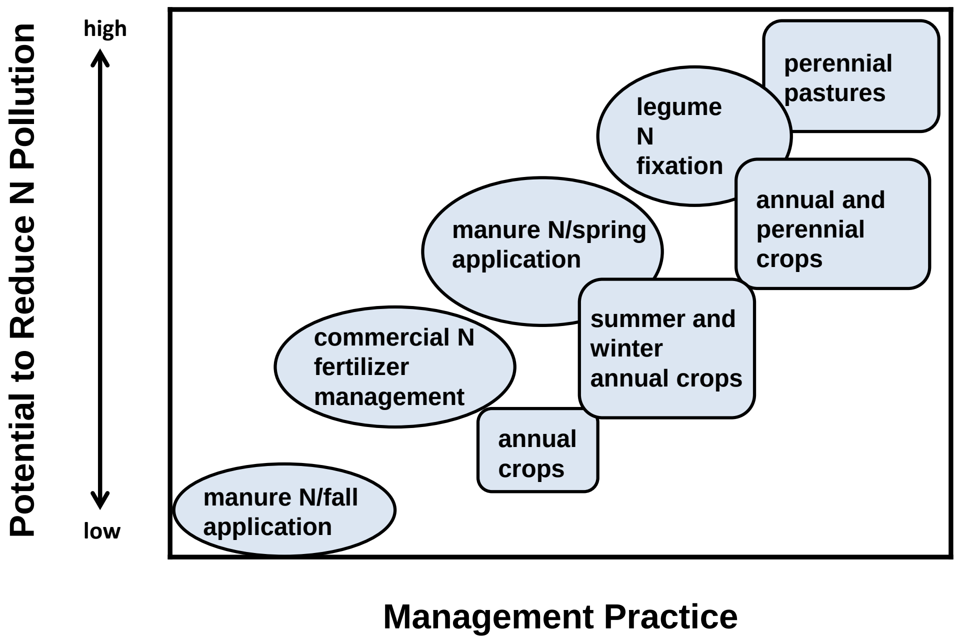

Figure 2.1 Ecological typology used to select farmers for this study based on management practices with differing potentials to reduce nitrogen (N) pollution of waterways. Different N sources are shown in ovals; different crop rotations are shown in rectangles (based on data from Blesh and Drinkwater, *in prep.*).

selecting 'alternative' farmers based on my natural science study, I avoided making simplifying assumptions about practices based on a broad category such as 'organic,' which is known to encompass wide variability in ecological impacts due to on-the-ground variability in how organic agriculture is actually practiced. By disaggregating problematic binaries such as organic/conventional, my work is based on a more subtle continuum grounded in ecological theory and measured outcomes.

Between 2008 and 2010, I conducted semi-structured interviews with 18 farmers in two contrasting regions of Iowa: the Des Moines lobe soil region of central Iowa and the Driftless Area of the northeast (Figure 2.2). These two regions have very different biophysical characteristics. Their glacial histories and geomorphic features affect present-day land use and the resulting environmental outcomes. The Driftless Area in the northeastern region escaped the most recent Wisconsinan-age glaciation, and has steep topography dominated by Alfisols, making it less suitable for capital accumulation through large-scale row crop production. In contrast, the Des Moines lobe in north-central Iowa is a flat glaciated landscape with younger, very fertile Mollisols. The installation of artificial subsurface drainage (i.e., "tile drainage") in fields in the region made the wet, high organic matter soils extremely productive (David et al. 2010). This region is highly industrialized and the landscape is dominated by large monocultures of corn and soybeans, which is reflected in the greater proportion of county land area in row crops and the greater total federal commodity subsidy payments (Table 2.1). Social processes co-evolve with these biophysical characteristics, and a cluster of innovation has formed in the more marginal environment of the northeastern part of the state. Though the two regions have similar access to state-wide agricultural and environmental organizations and programs through the Land Grant University (Iowa State University (ISU)), there are also important differences in their

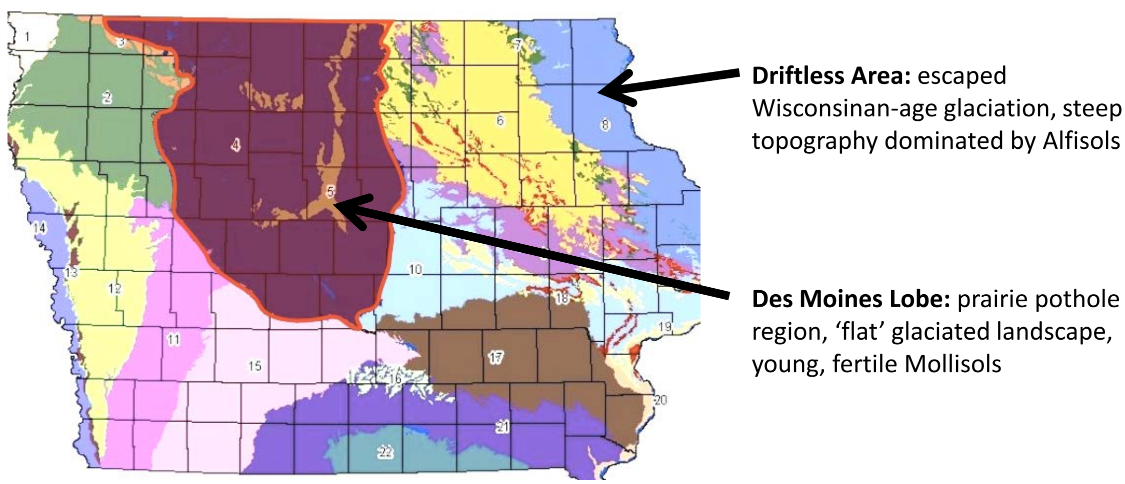

Figure 2.2. The two study sites were located in soil regions of Iowa with contrasting biophysical characteristics—the Des Moines lobe of central Iowa and the Driftless Area in the northeast.

respective agrifood system infrastructures; for example, the headquarters of the Organic Valley cooperative is located across the border from farmers in northeast Iowa in LaFarge, WI[3]. There is thus a well-developed alternative processing, marketing and distribution infrastructure in close proximity to alternative farmers in the northeast in sharp contrast to the Des Moines lobe region. There are also many more farmers who are certified organic and/or practicing rotational grazing on the "less ideal" agricultural landscapes of northeastern Iowa.

I selected farmer participants in each region systematically using *extreme case sampling* (i.e., in terms of the farming practices just described)—to locate "information-rich" cases that could provide detailed information about the transition process (Patton, 2001). The open-ended interview questions focused on eliciting the stories of how the transition to diversified organic grain production or intensive rotational grazing occurred. By focusing on *how* they did it, I also gathered information on the primary criteria affecting the decision for each farm household to transition. In the constructivist interview process knowledge production and interpretation are open; the narratives are "joint accomplishments of the coparticipants in communicative settings, and an intersubjective enterprise" (Oliveira, 2005; p. 423). Interpretive data are reconstructions of an original experience, in which it is important to consider my role as a researcher in co-producing and interpreting these data (Charmaz, 2003). My goal was, therefore, to collectively develop new understandings that impact the sustainability of agriculture in the region.

In my analysis, I aimed to both contextualize and conceptualize (Padgett, 2004), to develop a conceptual framework to understand the transition process in the upper Midwest, and to balance and link induction and deduction in an iterative process. I transcribed the interviews, which typically lasted between one and a half and two hours. To aid in conceptual analysis and

[3] This county also has the largest number of organic farmers in the nation (NOP, 2008).

Table 2.1. Select characteristics of the two contrasting study regions in Iowa.

	Soil Region	
	Des Moines Lobe	**Driftless Area**
Dominant soil order	**Mollisol**	**Alfisol**
Dominant soil texture	**silt loam**	**silt loam**
primary soil parent material	**glacial till**	**loess (eolian)**
native vegetation	**prairie**	**deciduous forest**
Corn acres (% of County land area)*	**46.4**	**30.7**
Soybean acres (% of County land area)*	**33.8**	**13.2**
Sum of corn and soybean acres (%)	**80.2**	**43.9**
Total Federal Commodity Payments**	**15,411,779**	**12,855,362**
Total Federal Conservation Payments**	**1,011,636**	**3,650,401**

*** Average of 2004-2008**

****Average of 1997-2006 in $/yr**

theory development, I drew on the constructivist grounded theory approach described in Charmaz (2003) to analyze the transcripts through thematic coding. In the analysis, I explored the resources and strategies that farmers used to navigate opportunities for, and rule out constraints to, transitioning to alternative practices. These were grouped into four broad categories: i) ecological and farm enterprise resources; ii) cognitive resources; iii) market opportunities; and iv) external/network relations with peers, knowledge organizations, and policy. The strategies drew upon both internal (i.e., personal or farm-level) and external (i.e., network) resources at multiple levels of organization. Through my analysis, I sought to elucidate tacit meanings which clarify farmers' perspectives and beliefs about reality. For practical reasons, I limited my methodology to farmer interviews, rather than including interviews with other important food system actors such as members of government agencies, environmental or agricultural organizations, or farm input suppliers. However, the narratives I elicited by emphasizing the "how" question provide insights into the opportunities and constraints to change at multiple levels of organization, and my analysis spanned micro- and macro-level factors. This work contributes to current understanding of transition dynamics on farms by privileging an integrated socioecological perspective that builds on an earlier ecological analysis of agricultural management practices, and, further, by focusing on *how* farmers transitioned to alternative production through a more symmetrical treatment of the role of structure and agency in the transition process.

Internal Resources

I. Ecological and Farm-Enterprise Resources

One strategy for successful innovation at the farm level was *harnessing biodiversity* on the farm. That is, in terms of identifying *who* is likely to transition in the current sociopolitical landscape, one thing these farmers had in common was either already having some level of ecological diversity on the farm or working to generate new diversity. For example, there might be some animals in addition to crops, or a small amount of hay ground. This biodiversity was a source of resilience to manage risk during the transition period: “That’s the beauty of having beef cows, weeds are still good. Now this is lambsquarters; you can cook this and eat it. There’s still nutrient value in weeds. I would mow it and bale it into hay. So the cows ate our mistakes” (Farmer 3). Another farmer told me about a failed organic soybean crop and said: “We got great hay off of it—hay that we could feed the pigs—so it wasn’t a complete loss” (Farmer 2). Thus making strategic use of existing diversity on the farm can help farmers stay profitable while learning to farm differently, highlighting the importance of ecological agency in shaping transition dynamics.

In many cases, these were smaller farms that had never been fully incorporated into the industrial system, and maintaining some diversity had been a strategy for keeping a small farm in the family. As one farmer put it: “we were already sort of different” (Farmer 2). A related motivation for transitioning to certified organic production or rotational grazing, especially for the diversified dairy farms in the Driftless Area, was young family members wanting to come back to the farm. One grazier said: “I just hope we can figure out how to diversify and do something different to be able to bring [my son] back to the farm, to figure out how to get more income” (Farmer 7). The younger generation had typically been exposed to new management ideas through college or farm internships, and families would exploit opportunities of alternative markets to generate sufficient income to support two families on a small farm.

Another farmer commented on the importance of scale by saying their farm was small enough that it is “still okay to cultivate a field or two. We haven’t sold the equipment for one thing. It’s a pretty easy entry for us, relatively speaking, compared to our neighbor farmers, who, you know, farm 3000 acres and who are not going to be pulling the cultivator out of a grove to do a small field” (Farmer 4). Thus, the barriers and opportunities to transitioning are relative to the respective starting point. And, in some cases farmers realized that by making minor adjustments to their operation—such as reducing chemical use and becoming certified organic, or beginning to rotate animals on pasture—they could take advantage of the price premiums of alternative agrifood markets to help maintain profitability on a small enterprise. In a region where competition for land and markets is extreme, this approach makes a lot of sense if the knowledge and equipment are still in place.

Complementing the importance of ecological or biological diversity on the farm was *developing enterprise diversity*, or *engaging in off-farm work* to supplement the income from farming. Rather than viewing these activities as indicating a ‘failure’ to succeed financially at alternative agriculture, we could view them as creative strategies for re-embedding agriculture into social and ecological spheres (e.g., Raynolds, 2000; van der Ploeg, 2010), within a political context that privileges the economic sphere. For instance, two sisters decided to pursue degrees in nursing in order to be readily employable in their small town and to provide extra income to support the family farm. Another farming couple ran a cage-free egg business along with their organic grain farm. There were farmers who planted large gardens and sold produce at the local farmers market for extra income. Other examples included farmers who invited visitors to the farm to participate in agri-tourism activities, or who ran a hunting lodge business for additional income. Before the industrialization of agriculture, farmers had no other option besides relying on ecological processes, but, today, creative household strategies that permit farmers to succeed at managing farms for multiple social and ecological outcomes are a prime location for farmer agency (van der Ploeg, 2010), and in making these choices, they are expanding the spaces for alternative production in the region.

II. Cognitive Resources

At the level of individual farmers, transitioning to new management systems also involves internal, psychological changes—such as thinking differently about the farm and farming. Farmers described new thought patterns and processes that reflected *ecological* or *systems thinking*; they variously identified and labeled these ways of thinking as “more holistic,” as “preventative thinking,” or as “working with nature.” For instance, alternative farmers cited weeds and soil fertility as their greatest management challenges. To successfully manage soil fertility in their alternative cropping systems, several farmers demonstrated in-depth knowledge of soil nutrient cycling processes, which reflected ecological thinking they had acquired to understand complex relationships between plant species, soil microorganisms, and ecological functions. One farmer, who had educated himself about the biological interactions in grain fields through extensive reading of soil science and agronomy literature, described the benefits of his ideal winter cover crop—a mixture of clover, vetch, and ryegrass:

> Well, you know that those roots are going to go down and all of those nutrients are coming up and they are being held in the organic matter instead of going down to the dead zone… And if you can let that cover crop get up to 6-8 inches, hold the nutrients until next spring, let it feed the bacteria, incorporate that into your top 3-4 inches [of soil] next year, it’s going to boost your organic matter, more nutrients available, more water holding capacity because you have more carbon there.” - Farmer 3

This farmer had a detailed understanding of soil nutrient cycling processes, and it is also interesting to note that he linked his comment to my research interests by referencing the Gulf hypoxia problem ("dead zone"), illustrating the intersubjective process of the interview itself—constructing knowledge through this joint conversation. Another farmer explained that he strives to achieve "as much of a closed loop as I can on the nutrient cycle" on the farm. The graziers were especially conscious of the fact that selling hay or forage represents a large transfer of nutrients off of the farm, and would avoid doing this whenever possible. And one grazier made the point that strategically purchasing hay is beneficial:

> It's a totally different mindset, I think. I don't want to put down anybody else, but you do have to look at things differently—renewability and recycling nutrients, and nutrient budgeting, for instance, purchased hay. You can think of it in terms of nitrogen, phosphorous, and potassium that you're bringing onto your farm instead of buying it in some factory or off the barge that came from who knows where. - Farmer 18

Even though farmers identified this shift in thinking as essential to the transition process, the distinction between an industrial "input orientation" versus an "ecological orientation" to management was not always so neat or linear. Not surprisingly, this distinction is better viewed as a continuum rather than a binary. There was a management gradient across interviewees in terms of relative reliance on purchased products versus building soil organic matter and "growing your own" fertility with legumes, for example. The dominant perspective among the farmers I interviewed was expressed well by this grain farmer, who eschews an input-substitution approach: "I guess that's been my philosophy. I read an article this winter and I thought it was very true, it said: 'do what's free first, what's cheap next, and if those don't work, then spend some money.'" But there were also some important counternarratives to the dominant 'ecological thinking' trend among the farmers I interviewed—some relied on purchased inputs, for example, such as manure from a neighboring conventional confinement operation, or OMRIapproved fertilizers from private service providers like Midwestern Bio-Ag[4]. Yet even these farmers identified problems or tensions with taking this approach. For example, one farmer who was having logistical challenges getting manure from his neighbor's farm to his farm was considering purchasing composted chicken manure from farther away and expressed his hesitation by stating: "It's kind of odd to be shipping in inputs on an organic operation." The following excerpt from another interview illustrates one farmer's process of shifting from an input substitution way of thinking to new ways of thinking about managing complex biological interactions and relationships on his farm:

> **Farmer 10:** One of the toughest things for me to manage has been soybean aphids in my soybeans. You know, I can spray for them on my conventional acres. The first year that we had really bad aphids, I don't remember if it was 2001, maybe, another organic grower and I bought 2 million ladybugs from some place in California in the mountains that grew ladybugs.
>
> **JBG:** [Laughing] How much does that cost?
>
> **Farmer 10:** It cost a couple hundred dollars and they came in these little sacks with moist wood shavings in them so that they'd have something to drink on the trip across country, and I went out walking through my soybeans just flinging ladybugs like this out on my soybeans, but the thing we didn't know is that, since they were raised in the mountains they wanted to go back to the mountains, and so they were there for a couple days and then all disappeared.
>
> **JBG:** They took off for the hills?!
>
> **Farmer 10:** [Laughing] Yeah! So, instead of staying at the buffet in my field where they could have all the aphids they wanted, they went looking for a climate more like theirs. And so they all disappeared. So, a couple hundred dollars—*a good tuition*. But now that we have soybean aphids, the ladybug population has naturally increased because there's so much more for them to eat, so it's not as hard to control them as it used to be. But that first year was interesting.

Ecological agency in the form of biotic relationships and feedbacks is important on all farms, not just alternative farms; the point here is that farmers identified that transitioning their farms to these new arrangements required shifting their thinking to pay more attention to, and explicitly manage, these relationships. The heightened attention to ecological thinking is important for success in an alternative management system because the short-term consequences

[4] http://www.midwesternbioag.com/

of biological interactions are more pronounced on low external input farms than in industrial production, where the primary management strategy is to override and control biological signals with chemical inputs.

In addition to demonstrating extensive knowledge of ecological relations, several farmers explicitly labeled their new ways of thinking as “independent,” or “not recipe farming,” recognizing the need to pay more attention to environmental variability on the farm, and to manage for that variability:

> With a conventional operation there are some pretty easy answers as far as what you do when you have [a specific problem]—you do this, this, and use this chemical, and they are pretty short-term answers, too. [In contrast], even with the professional advice we get on the organic side the answers aren’t perfectly straightforward. When you’re having certain weed issues, you know, you’ve got a weed seed bank in place in that field that you’re going to have to work through for a few years. - Farmer 4

Other farmers described this thought process as dynamic, rather than static, labeling it a “learning curve,” especially because every growing season is different. They identified that while they continued to rely on professional and outside resources after transitioning (referring to either private business or public actors such as the organic extension specialist at Iowa State University)—the *content* of the knowledge, and the types of interactions, had changed. The answers are no longer “easy,” and require a more explicit synthesis of place-based environmental particularities with external “professional” knowledge of agroecology: “There are a lot of good people and there is a lot of good information, it’s just not as straightforward as for the conventional knowledge world, where you just go to your input dealer, or your seed guy, or your fertilizer guy” (Farmer 4). Another farmer described this cognitive/behavioral transformation in the following way:

> I guess I’m sort of an introvert by nature, but to be successful with organics you really have to be more of an extrovert—to gain the information you need to do it—and so that’s a result of doing something really different like this. I guess that’s one way I’ve changed. - Farmer 5

Thus the internal and the behavioral changes that led to thinking more ecologically and systemically were mirrored in and cultivated by new *ways of relating* to others. The increased complexity involved in managing ecological interactions means that production information is most easily learned by conversations with many different people. These new ways of relating reflected a humility or openness in which farmers positioned themselves as learners rather than experts. For example, many farmers used the phrase "learning curve" to describe the process of acquiring the knowledge needed to successfully farm in a different way, and several farmers qualified their statements about this process by commenting "I'm always on the learning side of things" (Farmer 4). Another told me: "We're learning from our mistakes, too. Farmers make mistakes all the time" (Farmer 2). Even the experienced alternative farmers who frequently mentor others would describe the benefits and continued learning that result from these interactions.

Finally, farmers identified the development of new norms of interacting around market information as another behavioral strategy that involves a shift in thinking. Organic grain farmers, in particular, described the willingness within their new community of practice (Wenger, 1999; in this case, typically other organic grain farmers and alternative farmers) to share detailed information about markets, which had not been a social norm when they were farming conventionally. One farmer explained:

> We even share, you know, if you have a market for something that [another farmer] needs a market for, we'll give them market contact information. When I was farming conventional, I would talk to the agronomist at the local elevator, and that was about my main source of information. Now I use, you know, anybody that farms similar to us, and we all share information. - Farmer 1

These farmers suggested potential reasons for this difference, including different cultural values such as horizontal knowledge exchange, or the fact that organic grain farmers are rare and dispersed across the landscape of Iowa, and among this diffuse community there is less immediate competition for markets. One farmer commented that a couple of years before our conversation, which was in 2009, that it was “pretty easy to be enthusiastic about organic when the prices were so high” (Farmer 5), and this unsaturated and growing market perhaps partly explains the readiness to share market information in addition to management information.

III. Marketing Opportunities

Sourcing new markets is a completely different process for alternative farmers in the MRB than for industrial farmers and requires the development of new competencies and skills at the level of the farm enterprise. Both graziers and grain farmers spanned a wide range of strategies for marketing alternative products, from direct marketing to selling on commodity markets. Meat, compared to grain, is much more frequently sold in direct markets; however, there are several different types of grazing operations with different finished products. Cow-calf operations breed cows and sell the calves, often at auction but sometimes directly to neighboring farmers. There are “backgrounders” who bring weaned calves to “heavy feeder” weight (up to 800-1000 pounds, depending), and then “finishers” take the cows from heavy feeder to finish weight. Clearly the finishers are most likely to direct market meat to consumers. My sample of rotational graziers included cow-calf farmers and finishers, as well as dairy farmers who sold milk.

Among the graziers, farmers selected the type of grazing operation and markets that were appropriate for their skills, interests and personalities, a decision which interacted with their geographical location, particularly in terms of external opportunities such as access to a farmers market, nearby city, or processing infrastructure. For example, the graziers agreed that direct marketing meat can require enough labor to become a second job. There was a clear consensus that in the state of Iowa there is practically no local market for meat that is both grassfed *and* organic, and that regional or national markets for grassfed, organic meat are unstable. For example, in reference to Whole Foods, a farmer said: “Some of these guys who had been selling organic grassfed beef, the market kind of collapsed on them” (Farmer 11) explaining that Whole Foods had switched to buying from farmers in a different region of the U.S. Currently, there is little incentive in this region for beef graziers to certify their operations, though some chose organic certification for philosophical reasons. In contrast, for dairy farmers in the northeast there is a large incentive to become certified organic because of the opportunities provided by selling milk to Organic Valley, especially the stable price premium.

The cow-calf producers typically sold their animals at auction through the local sale barn where their animals were marketed as ‘green’ or ‘natural.’ One cow-calf farmer said: “We’re selling in the same market, but there are buyers that are looking at that niche of feeder calf, you know, that aren’t burnt out from being pushed so hard on creep[5] and all that stuff” (Farmer 9). Another farmer’s calves are “gold-tagged” at the sale barn, which means they are usually “at the top of the bunch, price-wise.” These calves eventually go to general slaughter, rather than to alternative markets, though they may stay in state—for example, the meat might be sold in Iowa at chain grocery stores like Hy-Vee. In the few instances in which cow-calf farmers were selling directly to alternative markets, particularly grassfed, there were challenges related to access and seasonality:

> Because it’s a niche market and they can only take so [many animals] at a time, we do have problems sometimes with moving the animals when we think they are ready. It’s different from the conventional market in that way—you know, they can only take so many at a time.

[5] Creep feeding refers to giving supplemental feed (usually concentrates) to a nursing calf. The feed is provided in a “creep feeder” or some type of physical barrier, which allows the calf access to the extra feed while preventing access to cows. Creep feeding nursing calves increases the rate of weight gain and the weaning weight.

> Conventional market, why, if you need to get rid of them you just take them to the barn and there's a sale once a week. - Farmer 14

Alternative grain farmers fell along a similarly broad spectrum in terms of the types of markets they were able to source or develop themselves. Rather than hauling grain to the local cooperative[6] or grain elevator, marketing alternative products such as certified organic grains involves extensive networking with potential buyers at farming conferences or other neighboring farmers, and, often, transporting the grain longer distances. Farmers typically go to conferences and network with millers or other brokers and tell them the quantity and quality of what they have to sell. Those millers will then call them and offer their current price when they are looking to buy grain. After the initial effort of making contacts, farmers told me these buyers usually stay in close contact and it is not necessary to continually seek new ones. Though it is a completely different process, for some farmers gaining these new competencies was fluid and smooth:

> So far, selling the grain has been the easiest part of organics—last year, I had a list of about five grain buyers who I had spoken to at conferences, so I just called them up and each one quoted me an on-farm pick up price, so it seemed pretty competitive, and then I just sold it to the highest bidder. They came with their trucks and picked it up. - Farmer 5

At the most alternative end of the marketing spectrum, a handful of grain farmers actually sold a small proportion of their product directly to consumers. For example, one farm couple direct markets popcorn and flax seed off the farm, and even engages in educational activities to teach potential customers how to prepare and eat their products. They have a booth at the local grocery store at which they describe the nutritional benefits of flax seed and flax meal and demonstrate how to use it in recipes. Another farm family grinds a small proportion of their grain into flour and bakes bread to sell at their town's farmers market. However, in general, grain crops are not biologically conducive to small-scale direct marketing. Such activities take a lot of

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[6] Industrial agricultural cooperatives, or "co-ops" are not member-owned cooperatives with pooled resources (a more typical meaning of the word "cooperative"), but instead refer to service providers for agricultural inputs, marketing or credit.

time; the benefits are largely educational, and have to do with building community support for alternative production and relocalization of agriculture. These grain farmers are vectors for change, but the largest proportion of their product—and income—comes from national- or global-scale markets.

Finally, alternative grain farmers also direct-marketed grains as feed to neighboring or nearby livestock farmers. This opportunity was greater for farmers in the northeastern region:

> There are enough people in our area that feed organic livestock that I've always been able to find a market for the corn. Always been able to find a market for hay because we have organic dairy producers here. My beans go to the soymilk project [through Organic Valley]. So, I haven't really had any trouble. - Farmer 10

In northeastern Iowa many dairy farmers sell certified organic milk to Organic Valley, and while they are required to provide animals with access to pasture, the cows are not strictly grassfed, and there is thus a large regional demand for organic feed grains. These marketing activities and relationships are actively creating new alternative agrifood system arrangements within the MRB. This new infrastructure, which is being built by farmers and other actors such as cooperatives and millers, is emerging in an iterative, co-evolutionary process. The dominant agricultural infrastructure of the upper Midwest does not embody the interests of these practitioners, so they are building something new and different as they go. This transition is also a hybrid of relying on, and slightly altering, the existing infrastructure, while simultaneously building a new infrastructure as the broad spectrum of marketing activities demonstrates.

External Resources

I. Relations with Peers: Farmer Networks

The internal, or farm-enterprise resources—ecological, cognitive, and the capacity to develop or source new markets—were interconnected with external resources such as network linkages. Opportunities for networking with peers were facilitated by alternative farmer organizations. These groups emphasize farmer-to-farmer interactions and relationships, and in the process have helped form new communities of practice centered on alternative and sustainable agriculture. There is a large literature describing the process of horizontal knowledge exchange that occurs through alternative farmer networks—for example at farmer field days, pasture walks, conferences, and through engaging in activities such as on-farm research. In addition to the importance of meetings and events, the organizations also publish material resources such as fact sheets, newsletters, and online resources like e-lists, websites and videos which farmers identified as important sources of information for successfully transitioning.

These networks serve multiple roles. They are important resources for gaining new management knowledge, but are also important sources of social support (Hassanein, 1999; Bell, 2004) for farmers whose agricultural practices are marginal, in an area where there is extreme social pressure against doing something different. After one Des Moines lobe grazier told me how isolated he feels being "stuck right out in the middle of corn and soybean country" he explained, "The first summer or two I was transitioning I [traveled] to a lot of pasture walks, and then my wife said to me, 'Well, don't you think you need to back off? Haven't you learned everything you can?' I said, 'Oh, I still like to go. That's kind of my vacation.'" Hassanein (1999) explored the role of farmer networks in the conversion experience for rotational graziers in Wisconsin, and noted that farmers exchanged knowledge about how to graze, "and in the process they also exchanged beliefs about why to graze" (p. 113). She noted the fluidity between the practical and ideological for farmers in the network. The farmer networks generated new, site-specific knowledge, and in the process of sharing this knowledge they forged connections to the wider sustainable agriculture movement and established an alternative knowledge system.

Supporting this literature on alternative farmers, the majority of the farmers I interviewed also relied heavily on farmer networks. The shared experiences in these networks were often expressed through shared language; for instance, to explain *how* the field days were important mechanisms for gleaning knowledge from others, farmers would say, “I ask them what’s worked and what hasn’t,” a phrase I frequently heard repeated verbatim. The importance of experiential, practitioner knowledge for successfully transitioning to alternative production cannot be overstated. This phrase embodies alternative agrifood discourses more broadly as well as farmers’ self-understandings about a critical role they see for themselves in transforming agriculture—sharing hands-on, practical knowledge in non-hierarchical exchanges that benefit both the teacher and learner.

These external networks and relations take on different forms in the contrasting biophysical regions of Iowa. Spatial proximity to a cluster of other alternative farmers in the northeastern region facilitates transitions, meaning that alternative production in the Driftless Area is no longer as much of an interstitial and socially isolating activity as it is in central Iowa. One farming couple in the Driftless Area told me that they don’t know if they would have transitioned at all if not for the community of organic farmers in their area: “if we’d had to go out and try to find somebody to help us it would have been tougher.” And, seeing successful examples of peers farming organically for quite a few years and seeing their crops and their farms: “had a big bearing on our decision” (Farmer 13). In contrast, for farmers in central Iowa, transitioning to alternative agriculture often means traveling greater distances to events such as conferences, or reliance on the internet. An organic grain farmer on the lobe explained how he learned to operate and repair the new machinery required for his organic grain operation by watching demonstration videos on You Tube. This same farmer also enrolled in an online course through Iowa State University on organic management. One central Iowa grazier, who is isolated from other graziers, remarked how excited he was when a couple of young farmers who wanted to graze cows moved to his area "because there wasn't anybody to talk to here before" (Farmer 7).

For the state as a whole, the grassroots farmer organization Practical Farmers of Iowa (PFI) has been extremely successful, and was cited by nearly all farmers as being an important resource in their transition process. Practical Farmers of Iowa was founded in 1985 with the pragmatic goal of reducing chemical inputs to farms while maintaining profits, a framing which encompasses a huge diversity of practices—from industrial farmers who reduce nitrogen fertilizer rates, to organic farmers, to management intensive rotational graziers. Hence the distinctly non-politically charged word "practical" (Rosmann, 1994) in the organization's name, which was carefully selected to avoid excluding farmers who are less ideologically-committed to transforming agriculture in Iowa. Farmers often cited membership in PFI as the primary motivation for transitioning. There was a range of participation in PFI activities, from solely attending the annual conference, to being an occasional or regular field-day attendee, to hosting field days and conducting on-farm research trials.

Practical Farmers of Iowa has formalized the "trial-and-error" learning approach common to many of the farmers interviewed by facilitating on-farm experimentation. The organization works with farmers to set up trials on their farms to develop place-based management strategies, and to compensate for gaps in public (e.g., Land Grant University) research on alternative production practices (Rosmann, 1994). One farmer told me: "PFI has very scientific field trials. You've got to have six replications and it's got to be random and stuff like that. I've done some of those with them." These activities were frequently cited as important for developing new information about management practices. In fact it is a cultural norm among PFI members to have test plots or trials on their farms (e.g., comparing different nutrient sources or tillage practices). One farmer described a project he developed comparing the economics of his organic operation with a neighboring conventional operation in a rigorous and systematic way. He decided to do this not strictly based on his interest in the economic question, but also because, as a PFI member, he felt pressure to engage in some sort of research but wasn't interested in the labor and time investment of managing test plots on his farm. He thought it would be simpler to develop an indoor and less time-sensitive project. This socialization role played by farmer organizations is also an important source of change in the region.

Besides PFI, farmers identified other grassroots farmer networks which also facilitate horizontal interactions, such as Holistic Management International (HMI), the Land Stewardship Project in Minnesota and the Organic Valley cooperative in Wisconsin. For example, a group of graziers in northeastern Iowa paid for a certified trainer from HMI to visit the area and lead a workshop:

> I organized an instructor to come two years ago now from Nebraska and he gave us a week long course on all of the different aspects of Holistic Management. He did financial for a day, and then grazing planning for a day, and biological monitoring. And then, there was [*sic*] one or two days in goal setting and then using the decision-making process. They've also got that social factor. - Farmer 14

This group of farmers then continued the conversation and the learning process through a discussion group hosted by different families on a rotating basis.

II. Knowledge Organizations

<u>USDA and Land Grant Universities</u>

Interviews with farmers in this study supported Allen's (2004) finding that the farmer-driven sustainable agriculture movement works on multiple levels when interacting with organizations and institutions that support agriculture: they are both creating new institutions, such as PFI, *and* making changes within existing institutions. These farmers and their networks are thus engaging in both 'bottom up' and 'top down' efforts to develop new infrastructure and competencies for alternative agrifood systems in the MRB. In some cases the top down efforts are also extremely effective. For instance, one USDA NRCS office in the Driftless Area has been home for several decades to a grassland specialist, Jim Ranum, who has a unique ability to identify with and engage farmers from across a broad spectrum of grazing philosophies. He has had a large impact in this region—practically every grazier in this study mentioned him as a valuable resource. In other cases, farmers identified the contradictory or ambiguous role that organizations play—especially ISU, the state's Land Grant. The dominant trend, historically and today, is that university research and education has paved the way for the industrialization of Iowa agriculture (Hinrichs, 2003; Vanloqueren and Baret, 2009). The farmers I interviewed expressed frustration that the Land Grant is failing to serve its public purpose, as this organic grain farmer noted: "We've got as many as 10-12 plant breeders working at one time on corn and soybeans [at ISU], and about three years ago, at least we had one on small grains, but now he left and they don't have anybody at ISU specializing on small grains" (Farmer 11). Then, referring to plant breeders, and agronomists in general, he said, "Their time is being bought up by Monsanto and lots of people [*sic*] like that."

In response to pressure from the sustainable agriculture movement, significant changes and countertrends *are* occurring within Land Grants, though, and ISU has developed strong programming in organic and sustainable agriculture. Many farmers mentioned the importance of the Leopold Center for Sustainable Agriculture, which is housed on the ISU campus. As one put it: "ISU was helped tremendously by the Leopold Center, which helped congeal a lot of this stuff" (Farmer 3). Many of the grain farmers in the study also cited the importance of the Organic Agriculture Program and Kathleen Delate, who is in an Organic Specialist extension position at ISU, for conducting relevant research and disseminating useful information (e.g., publications, and through the internet, courses, and field days). One of the main activities of the latter position is running a cropping systems experiment comparing different organic grain rotations and management systems. Several farmers mentioned this research as a useful source of management information, though one farmer recognized the limitations of university cropping systems trials conducted on agricultural experiment stations, which fail to reflect 'actually existing' social realities:

> All of that research is done on a quarter of an acre. There are umpteen grad students out there to pull weeds and do the 'cleaning' and I come out here and try to do it on several hundred acres and it's completely different. And now I've lived that, so I can transition ground better and faster than I could when I started. - Farmer 3

Many farmers provided similar examples of how they would blend their site-specific knowledge, which is situated in a particular ecological, sociopolitical and economic context, with the more general knowledge about alternative agriculture generated from university research. In addition, farmers were quick to note that the alternative agriculture movement is not advocating a return to some imagined or romanticized past but is creating new modes of production by blending current scientific and ecological knowledge with historical and experiential knowledge from working farms.

Private Service Providers

Iowa farmers identified the growing role of private service providers as resources for alternative agriculture knowledge and material inputs. Transitioning to ecological agriculture typically involves a shift in emphasis from knowledge specific to selling a particular product—an external input—to knowledge *as the product* itself, as one grazier noted:

> We've had a couple of different consultants out over time and they've helped us some, just giving us recommendations, but what is interesting is that *they are not really consultants that sell anything because we don't really buy anything*. I mean, we're just relying on the sunshine. They did get paid. But normally when you get a consultant they are selling seed or fertilizer or something. - Farmer 14

However, this perspective differed between the graziers and grain farmers, highlighting the importance of paying attention to material and ecological realities of different types of agroecosystems. For rotational graziers, a production system based on perennialization facilitates reliance on fewer external inputs compared to annual grain production systems. In contrast, the alternative farmers who purchase organic-approved inputs generally tended to rely heavily on the growing number of private businesses catering to certified organic farmers as a key resource for their transition process.

The distinction between these two extremes is often fuzzy, and interviews revealed nuanced perspectives on the role of private input suppliers among both graziers and grain farmers. Farmers had differing degrees of acceptance for the presence and influence of these businesses. Some exalted them as a key source of knowledge and inputs on their farm. One farmer said he relied heavily on Gary Zimmer's book, *The Biological Farmer*, for management information, and that "his company, Midwestern Bio-Ag, is the one I found at the Acres conference [the annual conference of the organization, *Acres U.S.A.*, which focuses on eco-farming], and so with their help, and with us studying, that's how we got more knowledgeable" (Farmer 3). Another farmer similarly relied on inputs from Midwestern Bio-Ag and told me: "They are very good agronomists and they know a lot about balancing of the soils, so I try to follow their recommendations" (Farmer 1). Also common among interviewed farmers, however, was skepticism of the validity of the claims and products of businesses aiming to profit from ecological agriculture. For example, a northeastern Iowa farmer, when describing meetings that are coordinated by a local alternative farmer organization, said:

> Sometimes they'll have a topic where an outside person will come in, and unfortunately it's usually somebody who wants to sell [an organic] fertilizer or something. And that's why, sometimes, I don't know if I want to bother going to the meeting because it's just going to be a sales pitch. I mean, there would be some information rolled into it, but basically all of this information is directed towards selling what this guy wants to sell because the last thing [he says] is: 'and here's what I have, that'll do everything that I just said, and it only costs $300 per barrel.' And it's like, o.k., so how much can I trust what you just told me? - Farmer 12

In contrast he finds pasture walks to be the most important source of knowledge for his farm, because these events are: "the best source, when you can just talk to another farmer about it, because he's not trying to sell you anything and he doesn't have anything to prove."

Commodity Organizations

Some of the alternative farmers were members of organizations associated with industrial agriculture in the region, such as the Farm Bureau, the Corn Growers Association, the Iowa Soybean Association, or the National Cattlemen's Beef Association. And several farmers had received degrees from 'typical' agronomy programs at ISU such as Farm Operations. These memberships and affiliations reflected a blending of knowledge from industrial and alternative knowledge worlds, which was a tool these farmers had harnessed to successfully transition their operations. One farmer, transitioning from conventional crop production to rotational grazing, still maintains some connections with his conventional co-op: "As long as you're buying some product from them they will do all of this forage sampling for you, and pay for it. And, I got a feed ration program from ISU" (Farmer 7). Another farmer explained this hybridity in the following way:

> We are somewhat dependent on [our conventional neighbors] in some ways, like the infrastructure we use sometimes, and we do need that. A lot of smaller farmers maybe don't have all their equipment, don't have a combine, they'll hire a conventional farmer to do that. Things like that. Or, they don't have the trucks or semis to haul stuff, which are already in the system. Those truckers are here because of the conventional market; they aren't here just for the organic farmers…in most cases, anyway. - Farmer 11

Thus the co-evolution of farmers and infrastructure in the MRB mirrors the way that grassroots farmer groups interact with agrifood institutions and organizations, in the sense that farmers rely on the infrastructure of the industrial system while also building new infrastructure.

III. Agricultural Policy

The alternative farmers in this study described fewer opportunities to transitioning to alternative agricultural systems at the level of national and international policy, likely due to power imbalances of the neoliberal food regime. I asked farmers to describe their history of involvement with Farm Bill programs. Many alternative farmers had been or were recipients of commodity payments, but were critical:

> It just seems like all these subsidies and stuff really aren't doing anybody any good. If you can't run a profitable business without subsidies, it doesn't seem like it's a profitable business, or it's not sustainable. It seems like a lot of the emphasis on most of the USDA programs are all for grain farming, but why? There's a large excess of grain. It doesn't make any sense. All the high fructose corn syrup and all kinds of stuff like that they didn't use to use…because they have all this corn, and the price is not what it should be. - Farmer 17

Several farmers described how shifting to organic production or rotational grazing diminishes the importance of commodity subsidies for farm income. When farmers receive a price premium for an alternative product (e.g., organic corn or grassfed beef), commodity payments become a smaller proportion of its total value. Further, once diversified, farmers have fewer acres in a commodity crop, and don't receive payments for hay or pasture ground, for instance. However, because direct and counter-cyclical subsidy payments are based on a farm's historical base acreage[7], there can be a delay before transitioning actually impacts subsidy payments; for

[7] Fixed payments tied to "base acres" originated with the 1996 Farm Bill for the following commodities: corn, wheat, grain sorghum, upland rice, and cotton. Soybeans, minor oilseeds and peanuts became eligible in the 2002 Farm Bill. Base acres of a certain crop, such as corn, were initially calculated based on the average acreage and yield of that crop between 1981 and 1985 and these payments have remained in subsequent Farm Bills. Yields and acreages were most recently updated with the 2002 Farm Bill. These

instance, several farmers who had transitioned former crop ground to pasture still receive subsidy payments on those acres as part of their corn base, and they will until the next time base acres are updated, potentially with each new Farm Bill. Thus some Iowa farmers made strategic use of their historical involvement with federal commodity programs as a resource for mitigating economic risk while transitioning to alternative production.

Alternative farmers in the MRB cited Farm Bill conservation programs as important in the transition process, though appropriation of federal dollars toward conservation is minor compared to the commodity title (e.g., Batie, 2009). The Food, Conservation, and Energy Act of 2008 (i.e., the 2008 Farm Bill) contained cost-sharing provisions specifically targeting organic farmers and graziers through the Environmental Quality Incentives Program (EQIP). For instance, graziers in both regions had received EQIP funding to support setting up water tanks, for establishing pastures and interior fencing for paddocks, rotating cows, and running water lines. The graziers in the northeast also benefitted from the exceptional grassland specialist at the local NRCS office, who had helped several farmers in the study to design and execute pasture and grazing plans. The spatial location of farmers also matters for farm conservation policy. Though graziers in both regions of Iowa cited the importance of EQIP payments, more organic grain farmers in the northeast described the importance of EQIP for their transition. When there is a critical mass of organic farmers in one county, for example, they can have a large influence on which practices NRCS (responsible for administering EQIP) chooses to subsidize because there is county-scale flexibility in terms of how federal conservation money is spent. The opposite is also true; an organic grain farmer who is on the Des Moines Lobe, and isolated from

payments are intended to provide flexibility; farmers are guaranteed a certain income for base acres covered commodities (including if it is fallow, but not for fruit or vegetable production), but can grow another crop if the price is high.

other alternative farmers, expressed frustration with the lack of access to EQIP payments for conservation because of his location.

The Conservation Stewardship Program (CSP), which one farmer called "the new kid on the block for government programs" (Farmer 9; though, it is actually a revised version of the 2002 Conservation Security Program) was also important to many farmers in the study for both grazing and organic grain production. Several grain farmers had taken advantage of CSP cost-sharing to seed a cover crop in the fall (with mixed results due to the unfavorable weather that year); and others had received payments for spring cover crops, nutrient management plans, and long-term rotations. When I asked farmers how policies could promote entry of other farmers into alternative production, nearly all of them mentioned enhancing conservation programs such as EQIP and CSP, and cost-share payments. They also proposed improving federal farm policy through innovative ideas including limiting the total annual federal subsidy payment, mandating a multi-year, diversified crop rotation (i.e., at least three crop species) to be eligible for any government payments, or establishing new metrics based on ecological knowledge—such as the proportion of the year with living roots growing in fields, or with hooves on the ground—as standards for conservation. Another farmer, who was also a former NRCS employee, described his ideal vision for developing more holistic legislation to cover all private lands on rural landscapes (i.e., not just farms) in the U.S., which would simplify conservation into one cost-share program (i.e., grounded in compliance and regulation) and one incentive program.

Conclusion

Adapting Wolf and Primmer's (2006) resource-based framework, I found that for alternative farmers in Iowa, reliance on resources that are internal and external to farms mediated agroecological innovation toward practices with the greatest potential to reduce nitrogen pollution of waterways. Successfully transitioning to new management systems requires navigating a constellation of opportunities and risks, and the development of new knowledge and skills to do so. Iowa farmers managed risk at the farm-level by increasing and managing biodiversity, and through pluriactivity—that is, either strategic use of off-farm employment, or developing enterprise diversity on the farm. Through a variety of resources, educational opportunities and relationships they developed new cognitive competencies, particularly thinking more holistically and ecologically—for instance, understanding and managing complex biotic interactions that drive ecosystem function. In addition to these important changes on farms, farmer transition dynamics were social and collective, being shaped by new ways of relating with peers, and by the evolving activities of knowledge organizations in the state and region. Farmer organizations have facilitated the formation of new communities of practice in the MRB typically by coordinating meetings, events and workshops that prioritize horizontal learning through dialogue and discussion. Iowa alternative farmers also highlighted the growing importance of sustainable agriculture research and education activities through ISU. Finally, alternative farmers identified increasing opportunities to take advantage of cost-share payments through federal Farm Bill working lands conservation programs.

The collective action of farmers and organizations, particularly in the northeastern region of Iowa, demonstrates that innovation is relational, and that actors can cooperate in participatory ways to manage resources more sustainably (Steins, 2002). As social learning occurs in these networks the dynamic terrain of ‘legitimate’ knowledge shifts and evolves, fitting with a social constructivist perspective, such that alternative farmers in the northeast are currently much less marginal than they were in the past, and are less marginal than alternative farmers in central

Iowa who are more isolated. Further, a relational ontological perspective helps us view the alternative agrifood systems of the MRB as networks of socio-technical and ecological relations. By challenging the society-nature dualism we can gain a better understanding of the non-humans that are also part of the network—including, for instance, the biophysical characteristics of the two regions like soil type and topography, the ecological interactions in farm fields, and the material resources generated by organizations and institutions.

These alternative agrifood opportunities and practices are emerging in a context of the economic logic of the neoliberal food regime. Not surprisingly, privileging the "self-regulating market" (Polanyi, 1944) via the commodification of nature has led to large social and ecological costs. Nutrient pollution of waterways is just one example of the alarming and far-reaching effects of human agricultural activities—a dead zone as large as the state of New Jersey in the Gulf of Mexico, thousands of miles from the primary source of pollution. The neoliberal responses to such global environmental crises of agriculture fail to acknowledge the importance of *where* and *how* pollution occurs; existing voluntarist conservation strategies do not target the most problematic regions of the MRB. However, I presented empirical evidence that even in Iowa, a state in the center of productivist agriculture, the situation is somewhat open—alternatives are rapidly growing from a small base, and knowledge of ecological agriculture is being cultivated 'in place,' supporting findings of other studies of alternative agriculture in this region (Bell, 2004; Brock and Barham, 2008; Hassanein and Kloppenburg, 1995; Hinrichs, 2003). Economic and politically-powerful interests may dominate the debate about farming and food, but there are important tensions, resistances, and pockets of innovation on the landscape.

Here I have argued that it is important to learn from, and expand spaces for, the bottom up change that is emerging in this region to address the socioecological costs of industrial food systems. There is value in taking the view from the margins seriously, and understanding the perspectives and practices of those with the least power, who are also crucial sources of innovation. Such “innovation niches,” which deviate from the rules of the dominant socio-technical regime, (Geels 2002, 2004; cited in Vanloqueren and Baret, 2009) help create a trajectory of change that can inform and potentially lead to future systemic transformations in this region. In addition to grassroots change, the sustainable agriculture movement in the Midwest also has top down impacts (Allen, 2004); for example, working with the state to advance alternative agriculture through new conservation programs, such as working lands programs. And farmers in this study identified the importance of a diverse range of institutions, federal conservation programs, and the sustainable agriculture and local foods social movements for their successful transition to alternative practices. For practical reasons, my methodology centered on interviews with farmers, though my analytical perspective places farm household transitions within a sociopolitical, economic and ecological context. The top down processes and strategies, particularly within agri-environmental institutions and organizations, and the ways in which farmers and these institutions co-evolve in the MRB, deserve further study (e.g., see Hufnagl-Eichiner, 2011).

Interviews with farmers revealed that *they saw* the greatest opportunities for change at the farm and farmer-network levels. Their agency and preference is reflected in, first, choosing to do something different and risky in this region and, second, in helping to build the knowledge base and infrastructure to succeed at alternative agriculture. These bottom up approaches largely bypass the state and could be considered interstitial innovation (Wright, 2010)—activities that take place in the spaces or “cracks” within a dominant logic or power structure. Some might argue that precisely because these alternative farmers are interstitial—occupying small niches in the MRB—that they are irrelevant, or that they potentially reflect what McCarthy (2005) has labeled ‘hybrid neoliberalizations.’ For instance, one could point to the lack of state regulation of non-point source pollution from farm fields and the devolution of this responsibility to local levels. Or, perhaps, to the fact that all farms in the MRB are nested within a globalized political economy of agriculture that transcends the state. Even the fact that success at alternative agriculture in this region relies largely on price premiums is problematic—particularly regarding social justice issues such as equitable access to nutritious, sustainably grown food. These dynamics potentially indicate that the sustainable agriculture movement is susceptible to incorporation and cooptation by agribusinesses and other powerful interests. Because these alternative agrifood systems simultaneously challenge *and* reflect industrial agriculture, do they unintentionally reproduce the problems of industrial agrifood systems?

I would argue that despite the reality of these contradictions and tensions, the bottom up sustainable agriculture movement in Iowa is in fact generating the foundation for more widespread resistance and socioecological transformation toward sustainable agrifood systems. What other starting point is there besides the present? There is, currently, no single alternative strategy, or set of strategies available that could immediately transform and sustainably replace the neoliberal agrifood system. As Wright (2010) points out, this is part of the definition of hegemony. He argues that interstitial strategies work by both altering the existing conditions for eventual social change, and slowly eroding and redefining the limits of the existing dominant system. For example, when cooptation of values occurs in some spaces (e.g., the growing presence of ‘industrial’ organic farms; Guthman, 2004) other farmers continue to evolve and generate new, creative fissures and spaces for alternative food production (Pratt, 2009). Farmers, and their networks and organizations continue to develop and refine knowledge systems and discourses that center on sustainable agriculture, through, for instance, meetings and events, material resources, and on-farm research programs. Taken together, these activities, actors, and actants comprise hybrid and dynamic sustainable agrifood networks in the MRB. Friedmann and McNair (2008) frame these interstitial activities as being part of a ‘Builder’ movement that is not necessarily confrontational. For example, the alternative farmers I interviewed work within existing markets but are also generating new types of markets, in addition to generating larger spaces for non-commodified relations in agrifood systems. These activities may eventually alter the power dynamics of existing institutional arrangements that make up the neoliberal food regime, in the Gramscian sense of creating a counter-hegemony. And there is an important epistemological component to these strategies. Though discourse is important in constituting current alignments of power (Foucault, 1980), sustainable agriculture discourses are also important for challenging the dominant logic and opening new possibilities for plural knowledges and values to shape the trajectory of agriculture in the region.

The opportunities and barriers experienced by farmers in this study differed dramatically in the two contrasting physiographic regions of the state. Northeastern Iowa is an area where diverse factors have aligned to foster innovation: the soil types and topography make large monocultures less feasible and the pressure from national-level drivers for commodity production are less extreme. Here, alternative farming is facilitated by access to a better developed infrastructure of organizations and resources that improve efficiencies in learning and mitigate some of the risks of transitioning. In terms of leverage points for change, we can learn from the practices of alternative farmers, and the interactions between farmers and the agrifood system infrastructure within these clusters, and work at multiple levels to expand this innovation. Farmers in both regions identified many examples of ecological factors that impacted their transition process. A more complete understanding of what is often parceled out as a “social” process—transitioning to alternative management—therefore requires knowledge of the ecological conditions and processes that are not, in fact, separate from the social realm. This study contributes a rigorous approach to defining which practices are ecological, while simultaneously acknowledging that all science is situated. As scientists we should therefore ask ourselves: what types of knowledge are we working to generate, and who does it benefit? By highlighting the strengths of the intersubjective and nuanced perspective gained from qualitative interviews, I hope to help expand spaces for other natural scientists to engage in similar work, reflecting the way that, together, diverse interstitial activities of alternative farmers in Iowa are generating new conditions for sustainable food production and consumption in the future.

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CHAPTER 3

HOW DO WINTER RYE COVER CROPS IMPACT THE CYCLING AND RETENTION OF ^{15}N-LABELED FERTILIZER IN AN ILLINOIS PRAIRIE SOIL?

Abstract

Human production of reactive nitrogen (N), particularly for agriculture, has led to consequences for air and water pollution. The root cause of N leakiness in agricultural ecosystems is uncoupling of carbon (C) and N cycles. Industrial agroecosystems are highly simplified, with shorter periods of living plant cover on the landscape. Use of winter annual cover crops has shown promise as a practice that reduces N losses in grain agroecosystems. We applied ^{15}N-labeled ammonium sulfate fertilizer at corn planting in May of 2009 at a long term corn and soybean variety trial on an Illinois Mollisol. We tracked the fate of the labeled fertilizer in treatments comparing winter rye (*Secale cereal*) cover to the typical winter bare fallow. We measured fertilizer recovery (absolute and as a proportion of added) in corn at harvest in the fall, and in rye biomass the following spring (May 2010), as well as in a range of heterogeneous soil organic matter (SOM) pools at both sampling dates. In the spring, total recovery of added ^{15}N in crops and soil was low, ranging from 37-45%. Due to unfavorable conditions for cover crop establishment, and an unusually rainy October, rye growth missed the optimal window for ^{15}N recovery and very little tracer ^{15}N was recovered in the rye. However, the cover crop significantly reduced soil inorganic N pools in the spring (11.1 kg N ha^{-1} in the bare fallow treatment compared to 1.9 kg N ha^{-1} in the cover crop treatment), by an amount similar in magnitude to total N uptake by rye biomass (23.7 kg N ha^{-1}), suggesting that a key role of cover crops is in scavenging inorganic N mineralized from the SOM pools. Experimental plots with greater corn biomass in the fall had greater total ^{15}N captured in labile SOM pools such as

dissolved organic N, microbial biomass N, and free particulate organic matter demonstrating the movement of ^{15}N from litter into soil. However, at both sampling dates the majority (75-78%) of the total ^{15}N recovered in the soil was in the large pool of organic matter mostly composed of humified material. This study reflected current challenges in establishing cover crops given the reduced window for cover crop growth in cropping systems with corn varieties that have longer growing seasons. We suggest that an important research and policy emphasis would be developing and optimizing management of grain cropping systems to include winter cover and reduced bare fallow periods.

Introduction

Human activities have fundamentally changed how elements cycle on a global scale. Crop productivity is frequently nitrogen (N) limited, and agriculture accounts for approximately 75% of the human-derived N globally (Vitousek et al. 1997, Galloway et al. 2008, Gruber and Galloway, 2008). The majority (~70%) of N produced for agricultural activities is from Haber Bosch fertilizer, with the remainder from increased cultivation of N-fixing crops (Galloway et al. 2008). Increases in reactive N production and the 'leakiness' of the N cycle have contributed to both air and water pollution problems (Vitousek et al. 1997, Galloway et al. 2008). For instance, nitrate (NO_3^-) is one of the top water pollutants in the world. One consequence of NO_3^- leaching to surface waters is eutrophication of coastal marine environments; there are now hundreds of hypoxic zones in coastal marine environments worldwide (Diaz and Rosenberg, 2008). In the U.S., N leaching to surface waters from grain farms in the Mississippi River Basin (MRB) is the primary cause of the hypoxic zone that appears annually in the Gulf of Mexico (Carpenter et al. 1998; McIsaac et al. 2001).

On average, 50% of applied fertilizer N is recovered by crops (Smil, 1999). Further, stable isotope (i.e., ^{15}N) experiments conducted in temperate grain cropping systems demonstrate that, on average, 38% percent of applied fertilizer is unaccounted for in crops and soil at the end of one growing season (Gardner and Drinkwater, 2009). The primary cause of N leakiness in agricultural ecosystems is uncoupling of carbon (C) and N cycles (Woodmansee, 1984; Drinkwater and Snapp, 2007). Most industrial agricultural systems receive pulse additions of N that exceed the capacity of plant and microbial assimilation, and they are thus N-saturated ecosystems (Aber et al. 1998; Fenn et al. 1998). In addition to receiving large inputs of very mobile N forms, industrial agroecosystems are highly simplified and have shorter periods of living plant cover on the landscape, further reducing potential C-sinks for inorganic N additions. For example, cover crops (crops not harvested) are rare in industrial grain rotations, and winter bare fallow periods typically last 4-8 months. Therefore, most industrial agroecosystems have reduced C inputs and stocks of SOM compared to diversified agroecosystems or natural ecosystems. They are thus characterized by reduced N immobilization and are subject to greater N losses (Drinkwater and Snapp, 2007). In the tile-drained soils of the upper MRB, NO_3^- leaching is a particularly important loss pathway for N (Gast et al. 1978; David et al. 2010), with the majority of losses concentrated into a short period during the winter and spring (McCracken et al. 1994; Royer et al 2006).

Cover crops are crops included in a rotation for different ecological functions such as reducing soil erosion, controlling weeds or pathogens, or reducing nutrient losses (Snapp et al. 2005). Cover cropping is an example agroecological practice that has the potential to reduce N pollution by immobilizing excess N fertilizer and retaining it in the system (Tonitto et al. 2006; McSwiney et al. 2010). In a recent meta-analysis of cover crop studies that measured NO_3^-

leaching, Tonitto and others (2006) reported that including cover crops in rotation with grains led to a 70% reduction in leaching, on average, compared to rotations with a winter bare fallow. Despite the promise of winter cover crops for reducing N pollution (e.g., Shipley et al. 1992; Ball-Coelho and Roy, 1997; Tonitto et al. 2006), more agronomic research has focused on adjusting the form, timing, rate and placement of N fertilizer rather than on adjusting crop rotations or cropping systems (Gardner and Drinkwater, 2009). As a result, rotations with cover crops are extremely rare on farm fields in the MRB (Singer et al. 2007). Further, there are currently social and economic barriers to cover crop adoption, such as establishment costs due to expensive seed, and opportunity costs of potentially impeding planting of the following cash crop (Snapp et al. 2005). Despite these risks, cover cropping has the potential to be rapidly adopted by farmers with significant changes in national farm policy because it can be incorporated into slightly modified versions of current industrial grain cropping systems.

There is a lack of data on cover cropping in the dominant corn-soybean farming systems of the MRB and even fewer data are available using ^{15}N methods. Stable isotope tracer experiments involve adding a small amount of ^{15}N-enriched substrate to label a source pool, such as fertilizer, and tracing the labeled N into sinks or loss pathways in the ecosystem. This method is particularly powerful for understanding the N cycle because different N sources can be distinguished (Stark, 2000). For instance, with an isotopic tracer the amount of recently-added N retained in the soil can be determined against the large background of soil N, and ^{15}N mass balances quantify the recovery of labeled N in plants and soil over time (Hauck and Bremner, 1976; Powlson et al. 1992). In a meta-analysis of 217 ^{15}N studies conducted in grain systems, we found that only four experiments planted a cover crop following a cash crop to trace the fate of newly added N in rotations with cover crops (Gardner and Drinkwater, 2009). Further, the amount of labeled fertilizer N recovered in the cover crop aboveground biomass is typically small. For example, in one of these studies, fertilizer uptake by a ryegrass cover crop was only 4 kg N ha^{-1} yr^{-1} out of 100 kg N ha^{-1} applied to the previous barley crop, yet the cover crop reduced NO_3^- leaching by more than 50% over two years (Bergström and Jokela, 2001). This raises biogeochemical questions about how cover crops improve retention of added fertilizer. The traditional view is that the primary mechanism is plant uptake of fertilizer N that would otherwise be available for loss during a bare fallow period (McCracken et al. 1994). However, given the small amount of labeled fertilizer recovered in aboveground cover crop biomass, it is possible that a more important short-term mechanism may be to enhance incorporation of N fertilizer into organic matter pools by supplying C for microbial N assimilation, thus building soil organic N pools over time and the capacity for internal N cycling to supply future crop N (Nissen and Wander, 2003). For example, a range of soil organic matter (SOM) pools contribute to plant available N, and managing soil N pools with different turnover times has been suggested as an approach to nutrient management that could lead to greater N use efficiency in agroecosystems (Drinkwater and Snapp, 2007). Particulate organic matter (POM) is a SOM pool that is particularly responsive to short-term management and indicative of soil quality (Boone, 1994; Carter, 2002; Wander, 2004). Light fraction POM (or free POM) is derived from recent plant and animal residue inputs with fast turnover times of weeks to years, whereas occluded POM is a pool of partially-decomposed litter and roots that are physically protected in soil aggregates (Wander, 2004). However, little is known about incorporation of added fertilizer N into these labile and more recalcitrant SOM fractions over time because the bulk of the ^{15}N research had focused on crop recovery of added fertilizer, or on the cycling of fertilizer between the very small pool of soil inorganic N and crops (Gardner and Drinkwater, 2009). One microcosm experiment in an Illinois Mollisol found that larger POM pools were positively correlated with ^{15}N retention in soil, suggesting that managing a range of SOM pools is a promising approach for improving cropping system N use efficiency and reducing reliance on fertilizer N inputs (Nissen and Wander, 2003).

We conducted a field-scale ^{15}N tracer study to improve understanding of N cycling dynamics in a tile-drained prairie soil in Illinois—one of several Corn Belt states that contributes disproportionately to N loading to the Gulf of Mexico (David et al. 2010). We aimed to address three main objectives with this experiment: i) to follow the fate of ^{15}N-labeled fertilizer into a range of heterogeneous SOM pools to improve our understanding of soil N cycling in industrial grain agroecosystems; ii) to quantify the extent to which a winter rye cover crop contributes to short-term N retention in corn-soybean rotation, and iii) to elucidate potential mechanisms of cover crop N retention by quantifying ^{15}N recovery in a range of labile to more recalcitrant soil N pools with and without winter rye cover. We hypothesized that labile SOM fractions, such as free POM, would have greater ^{15}N recovery and a greater proportion of total N from newly added fertilizer, compared to bulk soil. We also expected to find greater recovery of ^{15}N in plant biomass and soil in the rye cover crop treatment compared to the bare fallow treatment one year following ^{15}N addition. Finally, we expected that POM fractions in the cover crop treatment would have greater ^{15}N recovery compared to the bare fallow treatment because of the greater C inputs in the cover crop treatment, which could alleviate microbial C limitation thereby increasing N assimilation and incorporation of labeled fertilizer into SOM pools.

Methods

Site and soil

The experiment was initiated in spring of 2009 with corn planting at a long term corn and soybean research trial at the Maxwell Farm of the University of Illinois, Urbana-Champaign (40.063°N, 88.230°W), Department of Crop Sciences Research and Education Center. The experiment occupied 0.04 ha of the site. The primary soil is a Drummer-Flanagan silty clay loam (fine-silty, mixed, superactive, mesic Typic Endoaquoll). The soil texture is 20.7% sand, 40.8% silt, and 38.5% clay with a pH of 6.1. The cation exchange capacity was 26.1 cmol kg^{-1} soil, primarily saturated with Ca (57.6%), Mg (21.4%) and K (1.7%) (Penn State Agricultural Analytical Laboratory, State College, PA). The soil C content in the top 30 cm was 13885 ± 116 g C m^{-2} and the N content was 958 ± 9 g N m^{-2}.

Experimental design and site management

The experiment compared two treatments with four replicates in a randomized complete block design. The treatments were: i) corn fertilized with 150 kg N ha^{-1}, and no cover crop (winter bare fallow); and ii) corn fertilized with 150 kg N ha^{-1}, with a rye cover crop planted following corn harvest. There were eight large plots which were 16.3 m^2 each, and one 2.28 m^2 microplot was established in the center of each large plot. We also established 0 N controls (four additional large plots, also 16.3 m^2) with no N fertilizer added. Samples from the control plots were used as the reference for calculations of ^{15}N recovery. Corn (DeKalb DKC 61-69 hybrid) was planted on 23 May 2009. On 14 June 2009 unlabeled ammonium sulfate fertilizer was applied as a solid by hand to the area outside of the microplots for both treatments, in a band in the middle of corn rows at a rate of 150 kg N ha^{-1}. The microplots within the large plots were excluded from normal N fertilization and instead received ^{15}N-labeled ammonium sulfate (10.6 atom % ^{15}N), also at a rate of 150 kg N ha^{-1}, using the same application method as for the unlabeled fertilizer. Fields had been tilled with a chisel plow in fall of 2008, and were field cultivated in spring of 2009 before planting. Plots received P and K fertilizers based on soil tests. For weed control, plots received a pre-plant application of Lumax (atrazine + mesotrione + S-metolachlor) at a rate of 7.0 L ha^{-1}. Tefluthrin (a soil insecticide) was applied in furrow at planting at a rate of 0.11 kg a.i. ha^{-1}. Following corn harvest we broadcast common winter rye (*Secale cereal*) seed at a rate of 150 kg ha^{-1} into the large plots for treatment ii on 26 October 2009.

Corn sampling

Two whole corn plants randomly selected near the center of each microplot were hand harvested at anthesis (23 July 2009) and at harvest (5 October 2009) by cutting at the soil surface. Plants were separated into ears and stover, and were dried for 48 h at 60°C. After drying, the grain was shelled from the cob. All plant parts were weighed to determined dry matter (DM) yield, and dried grain and stover were ground coarsely using a hammer mill and grinder. They were then finely pulverized using a roller grinder. Cobs were finely ground in a liquid nitrogen CryoGrinder.

Rye sampling

Rye biomass was sampled on 13 May 2010 immediately before tillage and planting of soybeans. We sampled 0.25 m^2 of aboveground biomass from all rye microplots, cut at the soil surface. Biomass was weighed, dried at 60 °C for 48 h, and coarsely ground in a hammer mill, then finely ground in a roller grinder.

Soil sampling and analysis

We took soil samples at two time points during the experiment to correspond with corn sampling at harvest (5 October 2009) and with rye biomass sampling in the spring (13 May 2010). Twenty soil cores (2 cm diameter by 30 cm depth) were composited per microplot at each sampling, and we used the core method to measure bulk density. In each microplot, we took two additional soil cores (also composited per microplot) to 100 cm depth and separated each core into 0-30 cm, 30-60 cm and 60 - 100 cm increments. At both sampling dates, fresh soil was processed immediately for soil moisture and extractable inorganic N (NO_3^- and NH_4^+). Triplicate soil subsamples were sieved for inorganic N determination (to 100 cm depth) and for a 7-day anaerobic N mineralization incubation (0-30 cm samples only), both using a 2 *M* KCl extraction (Drinkwater et al. 1996). The amount of total NH_4^+ and NO_3^- in each sample was analyzed colorimetrically on a continuous flow analyzer (AlpKem, Ol Analytical, College Station, TX).

For the 0-30 cm depth increment we quantified a range of labile and recalcitrant N pools. Microbial biomass, macro organic matter (macro OM; organic matter larger than 0.5 mm) and rye live root biomass to 30 cm were processed within one week of sampling from soils that had been stored at 4°C. To measure dissolved organic C and N and microbial biomass C and N, triplicate, field moist soil samples were analyzed using the chloroform fumigation-extraction method (Brookes, 1985). Soluble C and N were extracted with 0.05 *M* K_2SO_4 from fumigated and non-fumigated samples by shaking for 4 h on an orbital shaker and filtering through nitrocellulose filter papers with 45 μm pores. Extracts were stored in the freezer until they were lyophilized and ground to a fine powder using a mortar and pestle for analysis. Microbial biomass C and N content was calculated by subtracting the N and C in unfumigated samples from the N and C in fumigated samples. We quantified macro OM to 30 cm for the fall samples, and macro OM and rye live root biomass to 30 cm for the spring samples, by modifying the wet sieving method described in Puget and Drinkwater (2001). In brief, triplicate subsamples of 125 g of fresh soil (for the spring samples we increased the subsample size to 250 g) were soaked overnight in approximately 200 mL of 10% sodium hexametaphosphate and were then decanted through 2 mm and 0.5 mm stacked sieves and washed under tap water to break up soil aggregates leaving behind very small stones and sand, and collecting roots and macro OM on the stacked sieves. For the spring samples, rye live roots on the 2 mm sieve were separated from the macro OM using tweezers; rye live roots were separated from the dead macro OM in the 0.5 mm fraction by transferring all of the material to petri dishes with a small amount of water and separating them in the lab using tweezers. All material was dried, weighed and finely ground in a ball grinder (rye roots and 2 mm fraction) or a roller grinder (0.5 mm fraction).

Remaining sieved and unsieved soil was air-dried for later analysis of other SOM fractions and total soil C and N. We separated light fraction particulate organic matter (POM; also called free POM, or fPOM), and occluded POM (oPOM) on triplicate 40 g subsamples using a size and density fractionation method (Marriott and Wander, 2006). The fPOM fraction is derived from recent root and litter inputs, is not associated with soil aggregates, and can represent a source or sink for inorganic N, depending on the biochemical composition and C:N ratio of the source material (Wander, 2004). This fPOM method for dry soil thus includes the macro OM, which was recovered from fresh soil, in addition to smaller POM that ranges in size from 250μm to 0.5mm. The oPOM is physically protected within soil aggregates, has typically undergone more decomposition than fPOM, has a lower C:N ratio, and is more likely to represent a net source of mineralized N (Wander, 2004; Marriott and Wander, 2006). The air-dried soil subsamples were gently shaken for 1 h in sodium polytungstate (1.7 g cm^{-3}) and allowed to settle for 16 h, after which we removed the fPOM floating on top of the solution by aspiration. The remaining heavy fraction, or oPOM, was shaken with 10% sodium hexametaphosphate to disperse soil aggregates and was then rinsed through a 53 µm filter. The material larger than 53 µm was retained, and the oPOM was separated from sand by decanting. Total C and N of fPOM and oPOM (to 30 cm) and total soil (to 100 cm) were measured on a Leco 2000 CN Analyzer (Leco Corporation, St. Joseph, MO).

Analytical methods

Once pulverized to a very fine powder, samples from all plant and soil fractions were weighed into tin capsules and analyzed for ^{15}N enrichment and total N content using a continuous flow Isotope Ratio Mass Spectrometer (Stable Isotope Facility, UC Davis).

The liquid inorganic N extracts were diffused to determine the ratio of ($^{15}NH_4^+$ + $^{15}NO_3^-$) to ($^{14}NH_4^+$ + $^{14}NO_3^-$) by modifying the acidified disk method described in Stark and Hart (1996). We simultaneously added approximately 0.2g of MgO and approximately 0.4g of Devarda's Alloy to the KCl extract solutions in small plastic specimen cups along with a strip of Teflon tape folded in half containing two sealed paper disks (punched from Whatman® No. 1 filter paper using a hole punch) that were acidified with 2.5 *M* $KHSO_4$. The extract solutions had been spiked with 25 µg N total (from stock solutions of $(NH_4)_2SO_4$ and KNO_3) because the NO_3^--N and NH_4^+-N concentrations in our extracts were very low. The cups were capped quickly and allowed to diffuse onto the acidified disks for 6 d at room temperature. We inverted all cups daily to prevent the formation of acid droplets on the sides of the cups. At the end of the diffusion the Teflon strips were removed and rinsed in deionized water. The disks were carefully removed with tweezers and dried in a desiccator before being analyzed for ^{15}N enrichment using a continuous flow Isotope Ratio Mass Spectrometer (Stable Isotope Facility, UC Davis).

^{15}N calculations and statistical analysis

Total recovery of added ^{15}N-labeled fertilizer was determined by ^{15}N mass balance. Absolute and proportional ^{15}N recovery in corn and rye biomass, rye roots, and in the measured soil fractions was calculated using an isotopic mixing model (Hauck and Bremner, 1976). The *% N from fertilizer* (i.e., the fraction of total N from labeled fertilizer) in each plant and soil pool was calculated as:

(1) $(\text{atom \% } ^{15}N_{\text{sample}} - \text{atom \% } ^{15}N_{\text{control plot}})/(\text{atom\% } ^{15}N_{\text{fertilizer}} - \text{atom\% } ^{15}N_{\text{control plot}})$.

The *total recovery* of labeled N fertilizer in a given pool was calculated by multiplying the proportion of N in a pool from fertilizer (equation 1) by the mass of N (kg N ha^{-1}) in that pool. The soil N pools that we measured have a range of turnover times from hours to decades, whereas the most recalcitrant N pools in soil can turnover on timescales of centuries up to millennia. Therefore, to calculate the partitioning of fertilizer N among all soil pools to 1 m depth we subtracted the amount of ^{15}N-fertilizer recovered in the fractions we measured—the sum of soil inorganic N, fPOM (which includes the macro OM fractions), oPOM, dissolved organic N (DON), and microbial biomass N (MBN)—from the total amount of ^{15}N-fertilizer recovered in the total soil N pool to 1 m. This pool is likely composed primarily of humified organic matter. We label this fraction as “humus” in the results section and figures even though humus is an operationally-defined pool, and this material likely includes additional components (i.e., which are not technically “humified”) such as organic matter bound to soil mineral components.

Statistical analyses were computed using JMP v.8 software (SAS Institute Inc., Cary, NC). Differences in fertilizer recovery in various N pools were compared for the two fertilized treatments using one-way analysis of variance (ANOVA) with cover crop as the main effect. In the results section the treatments are abbreviated as BF (bare fallow) and CC (cover crop).

Results

Corn yield

The summer season in 2009 was extremely wet and cool, followed by an unusually wet October (Figure 3.1), which meant that the corn grain matured slowly and harvest, on 5 October 2009, was later than usual. Overall, corn yields in the experiment were low, particularly for Illinois. We attribute the low productivity to adverse weather conditions. At anthesis and harvest, mean whole corn plant yields were greater with N fertilizer (Figure 3.2; 4.7 ± 0.3 (±SE) Mg ha^{-1} and 12.7 ± 0.6 Mg ha^{-1}, respectively) than in the unfertilized control plots (2.4 ± 0.2 Mg ha^{-1} and 5.7 ± 0.3Mg ha^{-1}, respectively). Due to the unfavorable conditions, the cover crop was not seeded until the end of October. Even though rye germination occurred before winter frost, no substantial growth of rye biomass occurred in the fall. The lack of rye biomass along with leaching conditions promoted by heavy October rains may have led to the loss of soluble labeled-N fertilizer before rye roots were able to capture it.

In addition to poor rye establishment, slight differences in corn biomass (Figure 3.2), and grain yield (Table 3.1), were measured in the bare fallow (BF) plots compared to the cover crop (CC) plots at corn harvest in the fall. None of the variables in Table 3.1 were significantly different between the BF and CC treatments; however, they were all slightly higher in the BF plot than in the CC plots. We cannot explain this trend (e.g., with soil characteristics that might correspond) and variation among plots should have been controlled for with the randomized

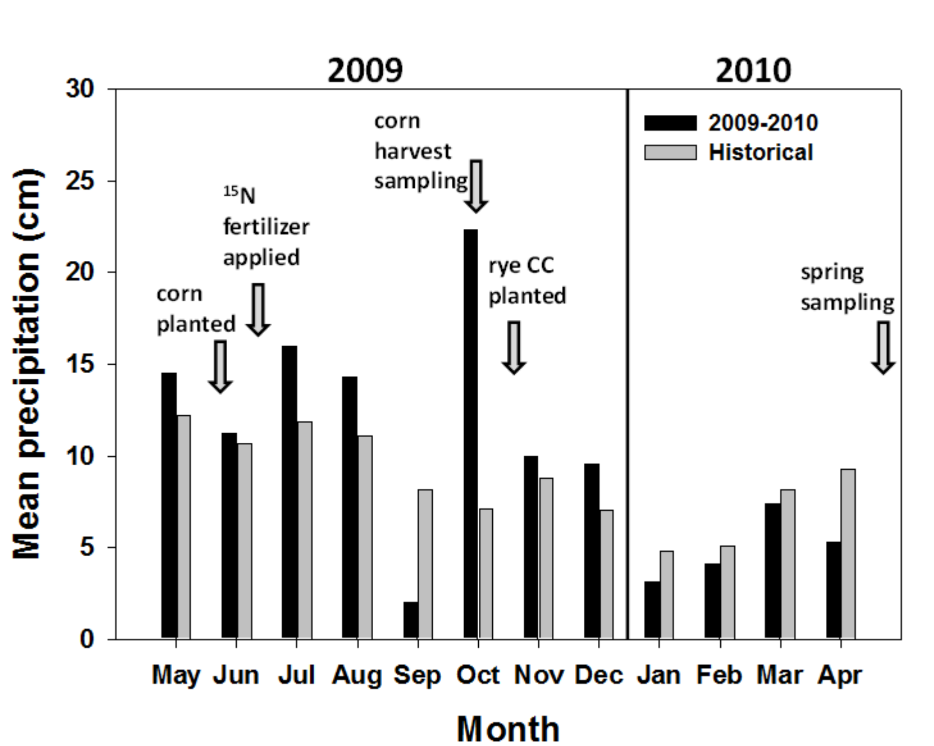
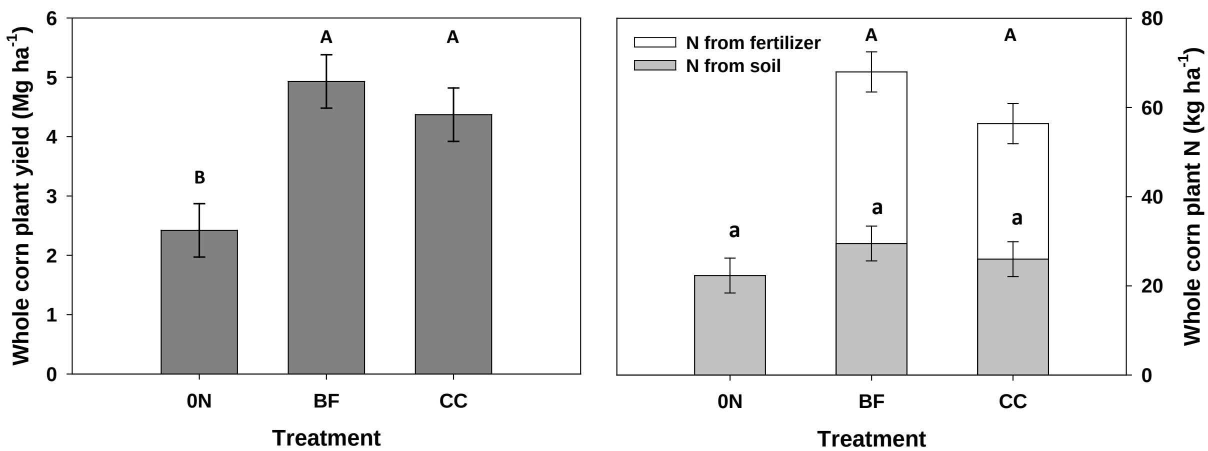


Figure 3.1. Average monthly precipitation (cm) during the experiment (2009-2010), and 30-year historical averages for Urbana-Champaign, IL.

A. Anthesis



B. Harvest

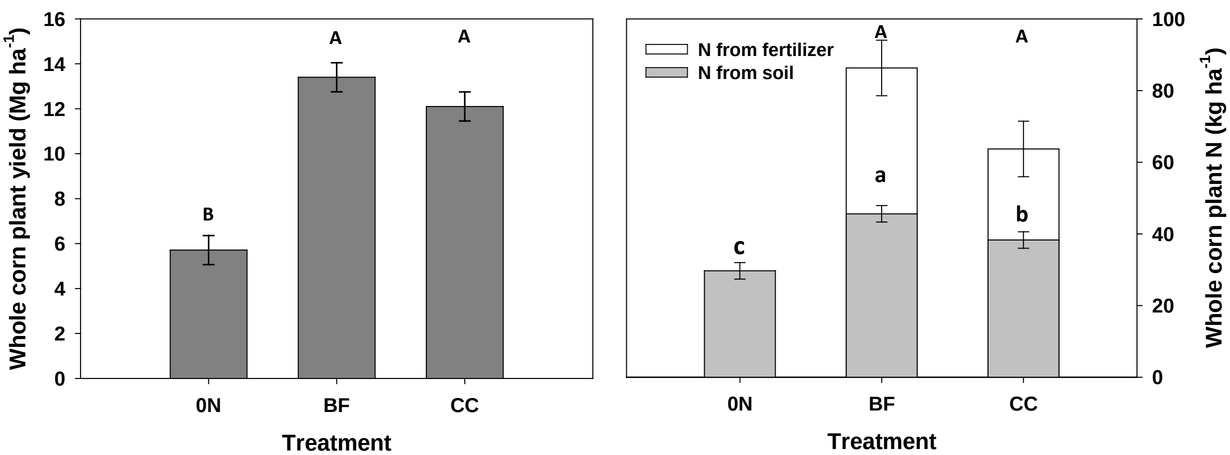


Figure 3.2. Whole corn plant yields (Mg ha^{-1}) and total N content (kg ha^{-1}) partitioned by source—labeled fertilizer N or soil N—for plants sampled at A) anthesis and B) harvest. Different letters denote significantly different comparisons ($p < 0.05$).

Table 3.1. Corn dry matter yield, total N content and N from labeled fertilizer. Observed means and standard errors (in parentheses), separated by plant part (grain, stover, and cob) at harvest. None of the variables were significantly different ($p < 0.05$) between the bare fallow and cover crop treatments.

	BF	**CC**
Dry matter (Mg ha^{-1})		
Grain	6.6 (0.7)	5.7 (0.1)
Stover	5.8 (0.3)	5.5 (0.1)
Cob	0.97 (0.1)	0.89 (0.1)
Total N content (kg ha^{-1})		
Grain	59.6 (9.5)	43.7 (2.4)
Stover	23.4 (3.2)	16.4 (0.3)
Cob	3.3 (1.0)	3.6 (0.3)
N from fertilizer (kg ha^{-1})		
Grain	29.0 (7.2)	18.7 (0.9)
Stover	10.3 (2.4)	5.2 (0.8)
Cob	1.5 (0.5)	1.4 (0.2)

complete block design since corn was planted and treated identically in all plots prior to imposing the winter cover treatment at corn harvest.

Fall ^{15}N recovery

A primary advantage of ^{15}N methods is that they allow fertilizer and soil N sources to be discerned. At harvest, uptake of soil N by whole corn plants (Figure 3.2B) was significantly different among all three treatments (p=0.001), and was greater in the two fertilized treatments (45.6 kg N ha^{-1} in BF and 38.3 kg N ha^{-1} in CC) than in the 0 N treatment (29.7 kg N ha^{-1}), indicating either a direct effect of fertilization or an indirect effect of plant growth priming SOM decomposition and N mineralization. That plant uptake of soil N was significantly different between the BF and CC treatments was problematic, since at that point no treatment had been imposed and variation among the plots should have been controlled for by the experimental design. Combining the two fertilized treatments (n=8) at harvest, the mean N from fertilizer in whole corn plants was 33.0 ± 5.5 kg N ha^{-1}, and the mean whole plant N uptake was 75 ± 7.5 kg N ha^{-1}. Thus, on average, 44.1% of corn N came from fertilizer and the rest was from soil, which is approximately the mean proportion reported by other corn tracer studies (Gardner and Drinkwater, 2009).

Grain yield was strongly correlated with the amount of labeled fertilizer that was recovered in the grain at harvest (Figure 3.3A; n=8), accounting for nearly all of the variation in fertilizer N recovery by corn. A weakly significant relationship between corn grain yield and labeled fertilizer recovered in the soil fPOM N pool sampled at corn harvest (p=0.09; Figure 3.3B) suggested that turnover of the fPOM pool is fast enough to reflect the impact of corn shoot and root residue inputs over a few-month timescale. This also confirms that recent plant litter inputs are an important source for and component of the fPOM pool. Even though grain was

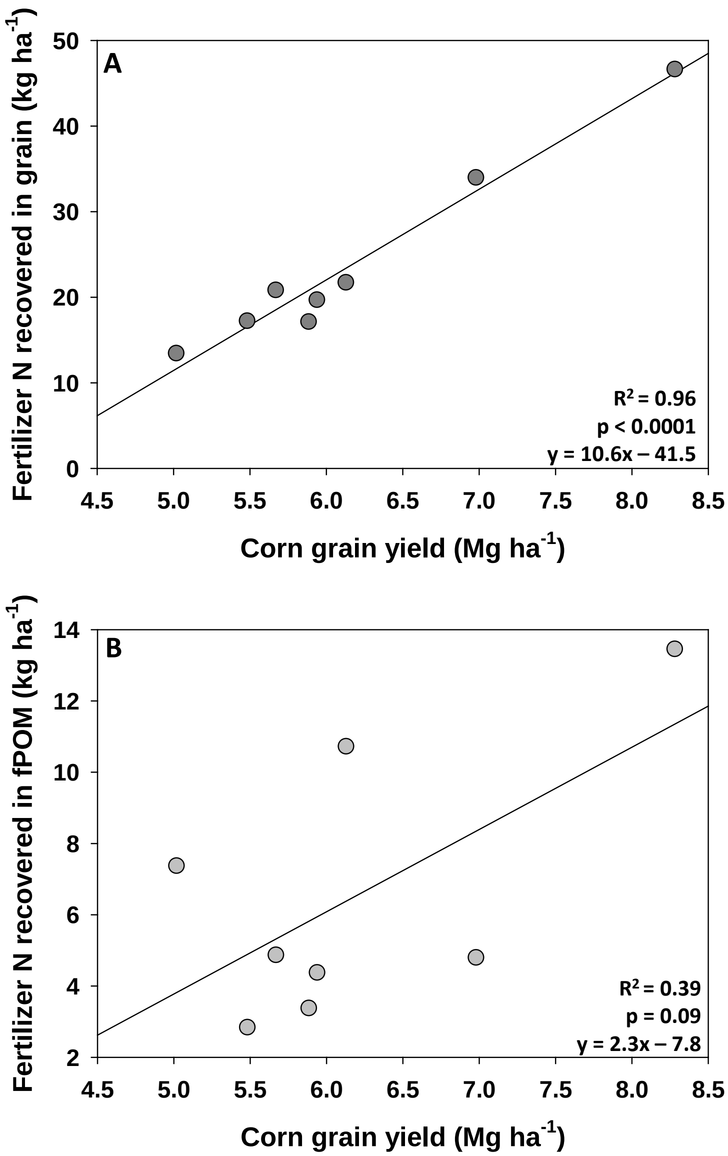


Figure 3.3. Relationships between corn grain yield and the total amount of labeled fertilizer recovered in A) corn grain at harvest (kg N ha^{-1}) and B) in soil fPOM (kg N ha^{-1}) at the harvest sampling.

removed at harvest, the impact of greater corn productivity and a trend toward greater ^{15}N recovery in the BF treatment carried through the experiment because close to half of the corn biomass (Table 3.1) remained in the field after harvest.

At the fall harvest sampling, total soil N content was significantly greater in the CC treatment (10037 ± 197 kg N ha^{-1}) compared to the BF treatment (9625 ± 253 kg N ha^{-1}) for the 0-30 cm depth increment (Figure 3.4A). There were no differences in total soil N for the other depth increments. Likely because of the slight differences in corn yield in the BF plots, there was a trend towards greater total fertilizer recovery in those plots compared to the CC plots on average (not significant; Figure 3.4B). The mean total fertilizer recovery in soil to 1 m was 50.9 kg N ha^{-1} in the BF plots compared to 43.8 kg N ha^{-1} in the CC plots (Table 3.2). However, in the 30-60 cm depth increment, mean fertilizer N was significantly greater ($p = 0.007$) in the CC (12.5 kg N ha^{-1}) treatment than in BF (4.1 kg N ha^{-1}) indicating that labeled fertilizer was already lower in the soil profile in those plots, and out of the plant root zone, before the cover crop was established. At the spring sampling (Figure 3.4D), the trend towards more fertilizer N in the BF treatment than CC treatment (27 v. 17 kg N ha^{-1}, $p = 0.2$) at 0-30 cm persisted, as did the trend toward more fertilizer N lower in the soil profile (60-100 cm) in the CC treatment (9.3 v. 5.0, $p = 0.2$).

Table 3.2 summarizes the total N content and the amount of ^{15}N-labeled fertilizer recovered (in kg N ha^{-1} and as a percentage of added fertilizer) for all of the measured pools at harvest: corn plants, total soil N to 1 m, and in the different soil fractions measured to 30 cm depth. Even though the BF and CC treatments had not been imposed yet, the results are separated by treatment to illustrate that the artifact of higher corn biomass in the BF plots lead to slightly greater fertilizer N recovery in labile soil pools in that treatment, where there should have been

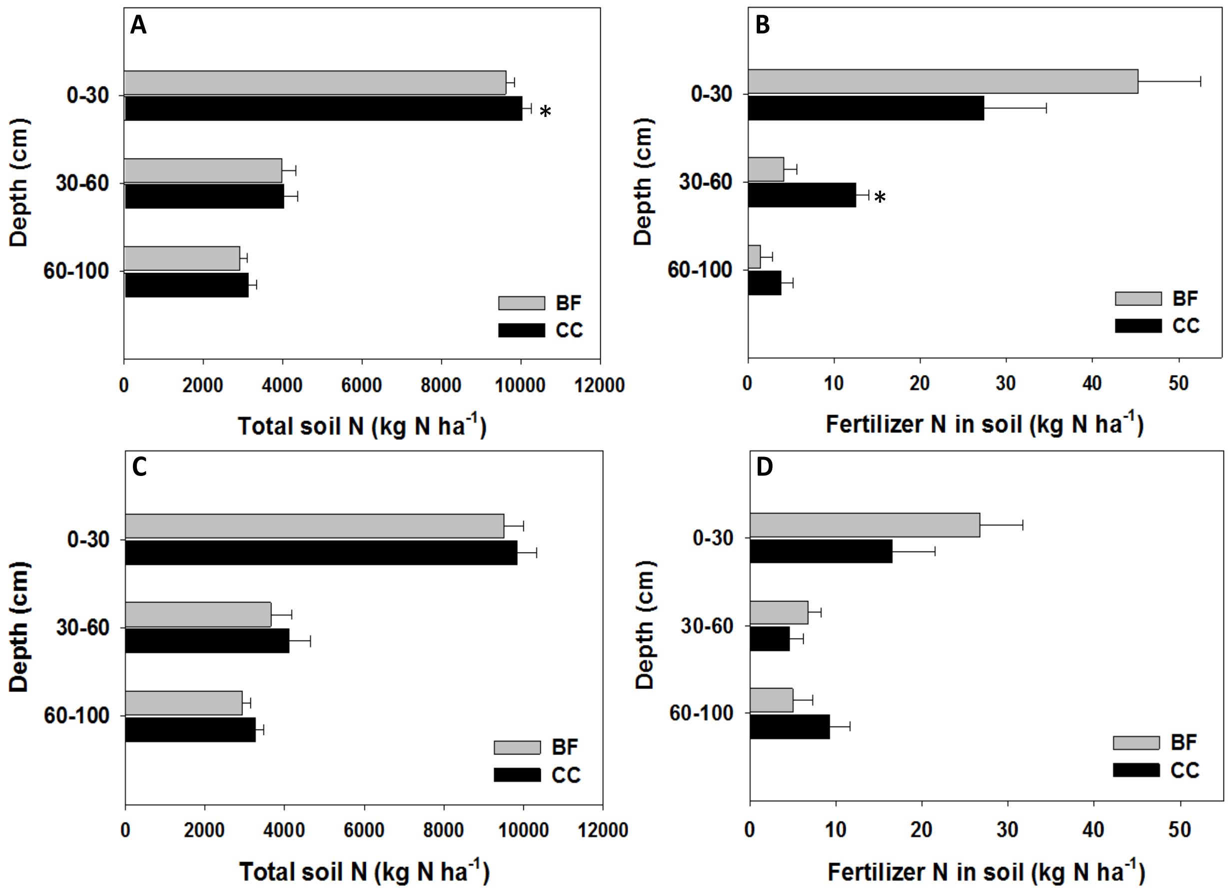


Figure 3.4. A) Total soil N (kg ha^{-1}), and B) fertilizer N recovered (kg ha^{-1}) by depth increment for the fall sampling on 5 October 2009 (* denotes statistically significant difference between treatments; $p<0.05$), and C) total soil N (kg ha^{-1}), and D) fertilizer N recovered (kg ha^{-1}) by depth increment at the spring sampling on 13 May 2010.

Table 3.2. Fall total N and ^{15}N distribution. Observed means and standard errors (in parentheses) for total N (kg ha^{-1}), labeled fertilizer N (kg ha^{-1}) and % recovery of added ^{15}N for the bare fallow (BF) and cover crop (CC) treatments before treatments were imposed at corn harvest on 5 October 2009. Statistically significant differences are denoted * ($p < 0.05$) or ** ($p < 0.01$).

	total N			fertilizer N			^{15}N recovery		
	kg N ha^{-1}			kg N ha^{-1}			%		
	BF	CC		BF	CC		BF	CC	
Crop									
Corn whole plant	86.3 (13.1)	63.7 (2.5)		40.8 (9.9)	25.3 (0.7)		27.2 (6.6)	16.9 (0.5)	
Soil									
total soil N (to 1m)	16519 (763)	17205 (454)		50.9 (9.6)	43.8 (3.6)		33.9 (6.4)	29.2 (2.4)	
Total crop + soil	16605 (751)	17269 (451)		91.7 (16.1)	69.1 (3.6)		61.1 (10.7)	46.1 (2.4)	
N in soil by fraction									
NO_3^--N + NH_4^+-N	12.4 (0.8)	10 (3.5)		0.31 (0.1)	0.31 (0.3)		0.21 (0.1)	0.21 (0.2)	
macro OM (2mm)	26.6 (7.4)	30.1 (4.7)		3.3 (1.0)	2.0 (0.4)		2.2 (0.7)	1.4 (0.3)	
macro OM (0.5mm)	46.2 (3.7)	54.6 (4.0)		4.0 (0.6)	2.5 (0.2)		2.6 (0.4)	1.7 (0.1)	
fPOM	115 (12.3)	92 (7.5)		9.1 (1.9)	3.9 (0.5)	*	6.1 (1.3)	2.6 (0.3)	*
oPOM	326 (35)	322 (31)		2.1 (0.3)	2.0 (0.5)		1.4 (0.2)	1.4 (0.3)	
DON	34.5 (1.4)	30.8 (0.8)	*	2.8 (0.6)	1.4 (0.6)	**	1.9 (0.4)	0.9 (0.4)	**
MBN	19.6 (1.6)	14.1 (1.2)	*	1.3 (0.2)	0.5 (0.2)	*	0.9 (0.1)	0.4 (0.1)	*

no differences. There was significantly more fertilizer N in the fPOM, dissolved organic N (DON) and microbial biomass N (MBN) pools in the BF plots than in the CC plots. There was no difference in the total soil inorganic N pool (NO_3^--N + NH_4^+-N) between treatments in the fall, and the amount of fertilizer N measured in this pool was very small (0.31 kg N ha^{-1}). There was also no difference in the anaerobic N mineralization potential measurement for the two treatments at the fall sampling (data not shown; mean of both treatments = 20.7 kg N ha^{-1}). There is some redundancy in the N in the measured POM pools. The fPOM method on dry soil includes the macro OM (larger than 0.5 mm), and POM that ranges in size from 250 μm to 0.5 mm. Recovery of labeled fertilizer in the fPOM pool was high (9.1 and 3.9 kg N ha^{-1}; Table 3.2), and was significantly greater in the BF treatment, indicating the movement of fertilizer N from corn litter into soil during the growing season. Total fall ecosystem ^{15}N recovery (plants and soil) was slightly greater in the BF treatment (91.7 kg N ha^{-1}; Table 3.2), but not significantly different from the CC treatment (69.1 kg N ha^{-1}). Accounting for the fertilizer N removed in corn grain at harvest, fall ecosystem recovery was 62.7 kg N ha^{-1} in the BF treatment and 50.4 kg N ha^{-1} in the CC treatment.

Spring ^{15}N recovery

At the spring sampling, immediately prior to soybean planting, mean rye shoot biomass was 1.2 Mg ha^{-1} and root biomass was 0.6 Mg ha^{-1} (Figure 3.5A). A very small amount of labeled fertilizer was recovered in the rye shoot biomass (0.53 kg N ha^{-1}) and root biomass (0.6 kg N ha^{-1}). The total N uptake by the rye cover crop was 17.7 kg N ha^{-1} in the shoots and 6 kg N ha^{-1} in the roots (Figure 3.5B). The rye roots were more enriched in ^{15}N (1.41 at% ^{15}N) than the shoots were (0.68 at% ^{15}N). Despite the poor cover crop growth, the rye scavenged soil inorganic N (Figure 3.6). There was significantly more NO_3^--N and NH_4^+-N in the bare fallow plots (7.3 v.

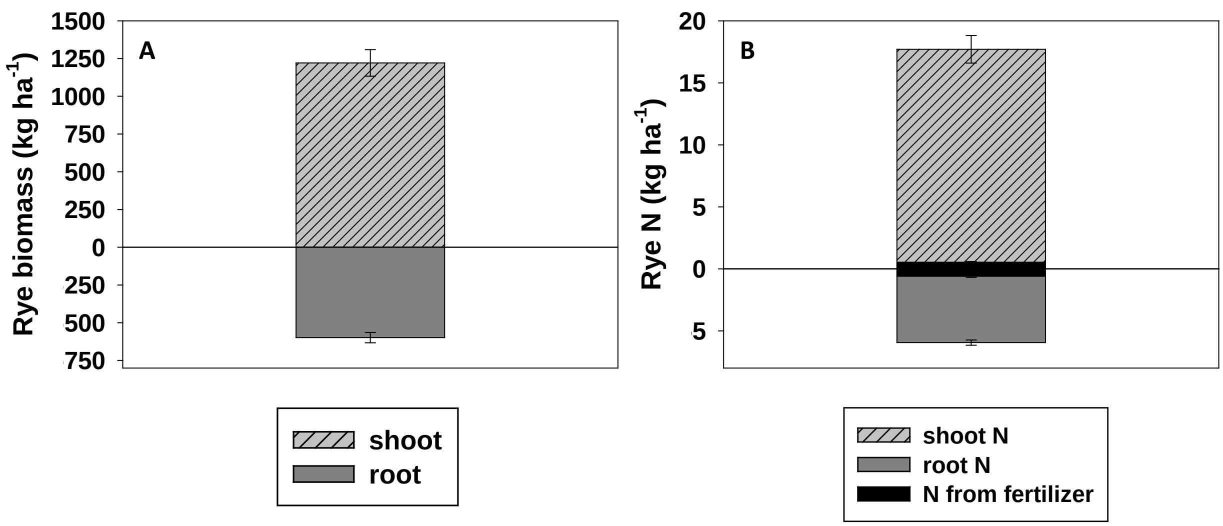


Figure 3.5. A) Mean rye shoot and root biomass (n=4) at sampling on 13 May 2010, and B) mean N content (kg N ha^{-1}) in the shoots and roots, and the amount of biomass N from labeled fertilizer.

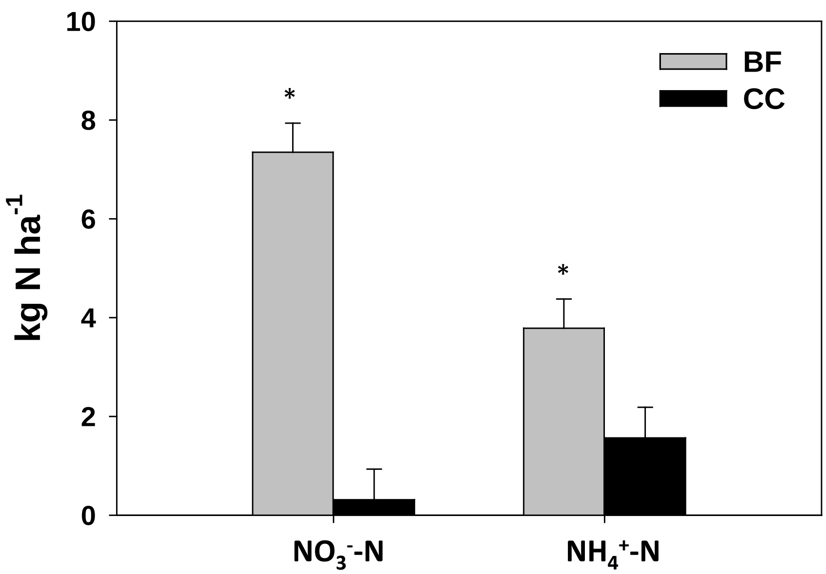


Figure 3.6. Total soil NO_3^--N and NH_4^+-N (kg N ha^{-1}) for the bare fallow (BF) and cover crop (CC) treatments sampled on 13 May 2010 (*denotes statistically significant difference between treatments; $p < 0.05$).

3.8 kg N ha^{-1}, respectively; p=0.002) in the spring than in the cover crop plots (0.3 v. 1.6 kg N ha^{-1}, respectively; p=0.038). Further, there was a negative relationship between rye biomass and the soil NO_3^- pool for the individual microplots with rye (Figure 3.7; p=0.09, n=4). The amount of ^{15}N recovered in either the NO_3^--N or NH_4^+-N pool for both treatments was practically 0 (Table 3.3).

In the spring, the absolute amount of fertilizer N recovered in soil and the % recovery of added ^{15}N fertilizer to 1 m depth was significantly greater in the BF treatment compared to CC (Table 3.3; p =0.04). Total ecosystem recovery in the spring (soil plus cover crop) was 38.5 ± 6.1 kg N ha^{-1} in the BF and 31.5 ± 6.6 kg N ha^{-1} in the CC treatment (p = 0.06). The mean N mineralization potential (from the anaerobic incubation) was also significantly greater in the BF treatment (45.5 kg N ha^{-1}; p=0.01) than in the CC treatment (21.3 kg N ha^{-1}). The fertilizer N content and the % recovery of added ^{15}N in the 0.5 mm macro OM was significantly greater in the BF than in the CC treatment (Table 3.3; p = 0.01). Similarly, the fertilizer N and % recovery of ^{15}N in DON and MBN pools were also significantly larger in the BF than CC treatment (p =0.01 and p=0.03, respectively). In addition, the total DON pool size was larger in the BF treatment (p =0.03). The atom % ^{15}N enrichment of the MBN pool was also different between treatments at the fall and spring sampling times (Figure 3.8; p=0.005 for fall and p =0.08 in spring) and declined from fall to spring, indicating movement of labeled fertilizer N out of this pool over time.

Since we found differences between treatments in the fall, when there should have been none, we normalized the spring results based on fall recovery (i.e., to examine treatment differences based on the change between fall versus spring), but that did not change the results of the statistical analyses in Table 3.3. For example, as a percentage of fertilizer recovered in the

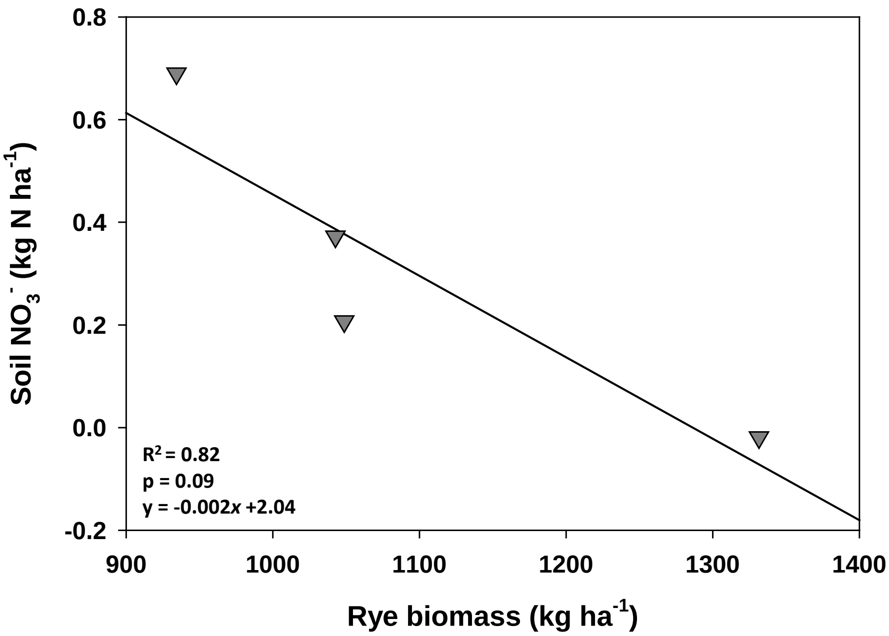


Figure 3.7. Relationship between rye biomass (kg ha^{-1}) and soil NO_3^- (kg N ha^{-1}) measured at the spring sampling on 13 May 2010.

Table 3.3. Spring total N and ^{15}N distribution. Observed means and standard errors (in parentheses) for total N (kg ha^{-1}), labeled fertilizer N (kg ha^{-1}) and % recovery of added ^{15}N for the bare fallow (BF) and cover crop (CC) treatments at the spring sampling on 13 May 2010. Statistically significant differences between treatments are denoted * ($p < 0.05$) or ** ($p < 0.01$).

	total N			fertilizer N			^{15}N recovery		
	kg N ha^{-1}			kg N ha^{-1}			%		
	BF	CC		BF	CC		BF	CC	
Cover crop									
rye shoots		15.5 (1.1)			0.53 (0.05)			0.35 (0.04)	
rye live roots (0-30 cm)		5.9 (0.2)			0.6 (0.1)			0.4 (0.1)	
Soil									
total soil N (to 1 m)	16082 (477)	17187 (1540)		38.5 (6.1)	30.4 (6.6)	*	25.6 (4.0)	20.3 (4.4)	*
N in soil by fraction									
NO_3^--N + NH_4^+-N	11.1 (1.5)	1.9 (0.5)	**	0.01 (0)	0 (0)		0 (0)	0 (0)	
macro OM (2 mm)	22.9 (5.9)	26.5 (3.8)		1.6 (0.3)	1.4 (0.2)		1.1 (0.2)	0.95 (0.1)	
macro OM (0.5 mm)	60.6 (2.6)	53.6 (5.4)		4 (0.5)	1.8 (0.4)	*	2.7 (0.3)	1.2 (0.3)	*
fPOM	71.3 (8.3)	96 (19.1)		4.3 (0.9)	3.8 (0.7)		2.9 (0.6)	2.5 (0.5)	
oPOM	293 (14.0)	287 (28.9)		2.7 (0.3)	1.7 (0.6)		1.8 (0.2)	1.1 (0.3)	
DON	39.6 (2.5)	30.5 (1.6)	*	0.8 (0.1)	0.2 (0.1)	**	0.54 (0.07)	0.15 (0.03)	**
MBN	17.7 (2.6)	7.2 (2.5)	p = 0.06	0.4 (0.1)	0.1 (0.06)	*	0.24 (0.1)	0.06 (0.02)	*

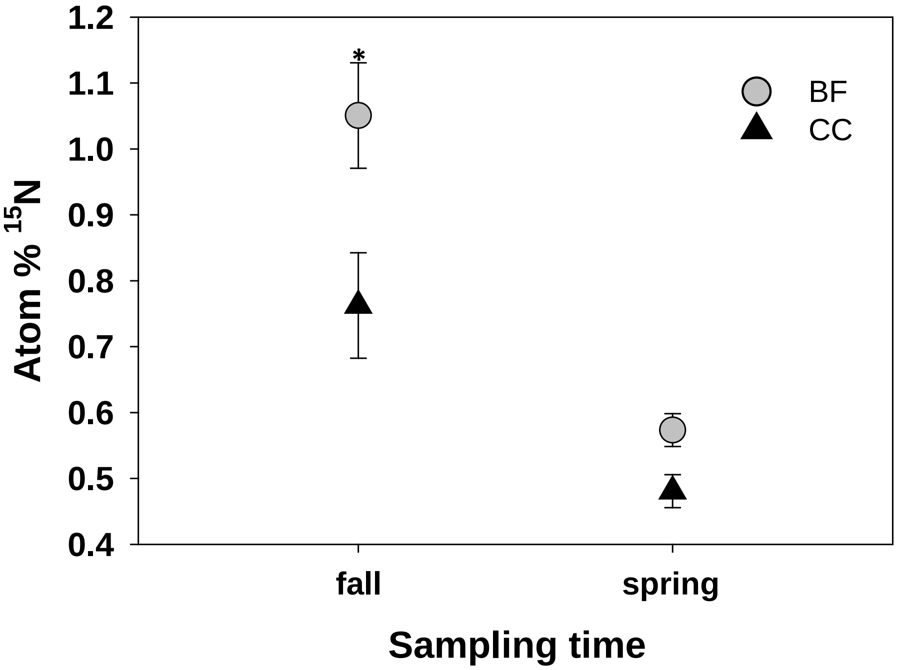


Figure 3.8. ^{15}N enrichment of the microbial biomass N pool at the fall and spring sampling times for the bare fallow (BF) and cover crop (CC) treatments (* denotes statistically significant differences between treatments; $p < 0.05$).

soil to 1 m in the fall, 76% was remaining in the soil in the spring in the BF treatment, compared to 70% in the CC treatment. Further, the difference in fall versus spring ecosystem recovery (i.e., the mass balance for plants and soil, accounting for ^{15}N removal in grain at harvest) was not significantly different between treatments, though it was slightly larger in the BF treatment (24.3 ± 12.4 kg N ha^{-1}) than in the CC treatment (18.8 ± 5.6 kg N ha^{-1}). Thus, even accounting for the different starting points in the fall, the magnitude of total fertilizer N loss over the winter was not significant between treatments.

Temporal dynamics of ^{15}N in soil

Though we found significant differences between treatments for several soil pools measured at both the fall and spring sampling dates, we averaged the two treatments together to explore the temporal dynamics of ^{15}N fertilizer movement in the soil (Figures 3.9-3.11). The % of added fertilizer that was recovered in the soil declined for all soil pools between the fall and spring sampling times, except for fertilizer N in the oPOM fraction, which stayed the same (1.4%; Figure 3.9). This indicates that N was moving out of pools with rapid turnover times, such as MBN, DON and fPOM. We also found that the amount of N fertilizer recovered in the fPOM pool at the fall sampling date accounted for 33% of the variation in the amount of fertilizer N recovered in the oPOM pool in the spring, though the relationship wasn't statistically significant (data not shown; p=0.13). The decline in the % recovery of added fertilizer in the soil over time also indicates that N was lost from the soil during the bare fallow period. The greatest proportion of added fertilizer N that was recovered in the soil was in the humified OM pool at both sampling dates.

In addition to the percent recovery of the labeled fertilizer N in soil (Figure 3.9), it is interesting to examine how the fertilizer N was distributed among the different soil fractions (i.e., as a percentage of the total N fertilizer that was recovered in the soil; Figure 3.10). In the fall, 47.4 kg N ha^{-1} of fertilizer was recovered in the soil to 1m (or 31.6% of added ^{15}N) for the BF and CC treatments combined, and in the spring, 34.5 kg N ha^{-1} was recovered (or 23.0% of added ^{15}N) for the combined treatments. Similar to the results for the % recovery of added ^{15}N-fertilizer (Figure 3.9), the greatest proportion of the fertilizer N recovered in soil was in the 'humus' fraction in both fall and spring. However, the humified OM pool accounted for a slightly greater proportion of the fertilizer N in the soil in the spring (i.e., of the 34.5 kg N ha^{-1} remaining in the soil in the spring, 80% was in the humified fraction) compared to the fall, indicating that N loss occurred preferentially from the more labile N pools. In terms of the distribution of fertilizer N in the soil, the oPOM pool accounted for a greater proportion of fertilizer N in the soil in the spring compared to the fall (Figure 3.9), likely because oPOM is also a recalcitrant pool of OM.

Finally, for each sampling date we calculated the percentage of each SON pool that was comprised of fertilizer N. Fertilizer N did not make up a large proportion of any of the pools (i.e., none were greater than 7%; Figure 3.11). Even though the greatest proportion of the total ^{15}N fertilizer recovered in the soil was found in the humified fraction (Figure 3.10), N fertilizer accounts for an extremely small percentage of the total amount of N in that pool (i.e., isotope methods enable detection of the labeled fertilizer even though its mass is extremely small compared to the total amount of N in stable SOM pools). In the fall, fertilizer N comprised a greater proportion of the labile pools—soil inorganic N (SIN), MBN, fPOM, and DON—than it did in the spring (Figure 3.11), indicating movement of fertilizer N out of these pools over time.

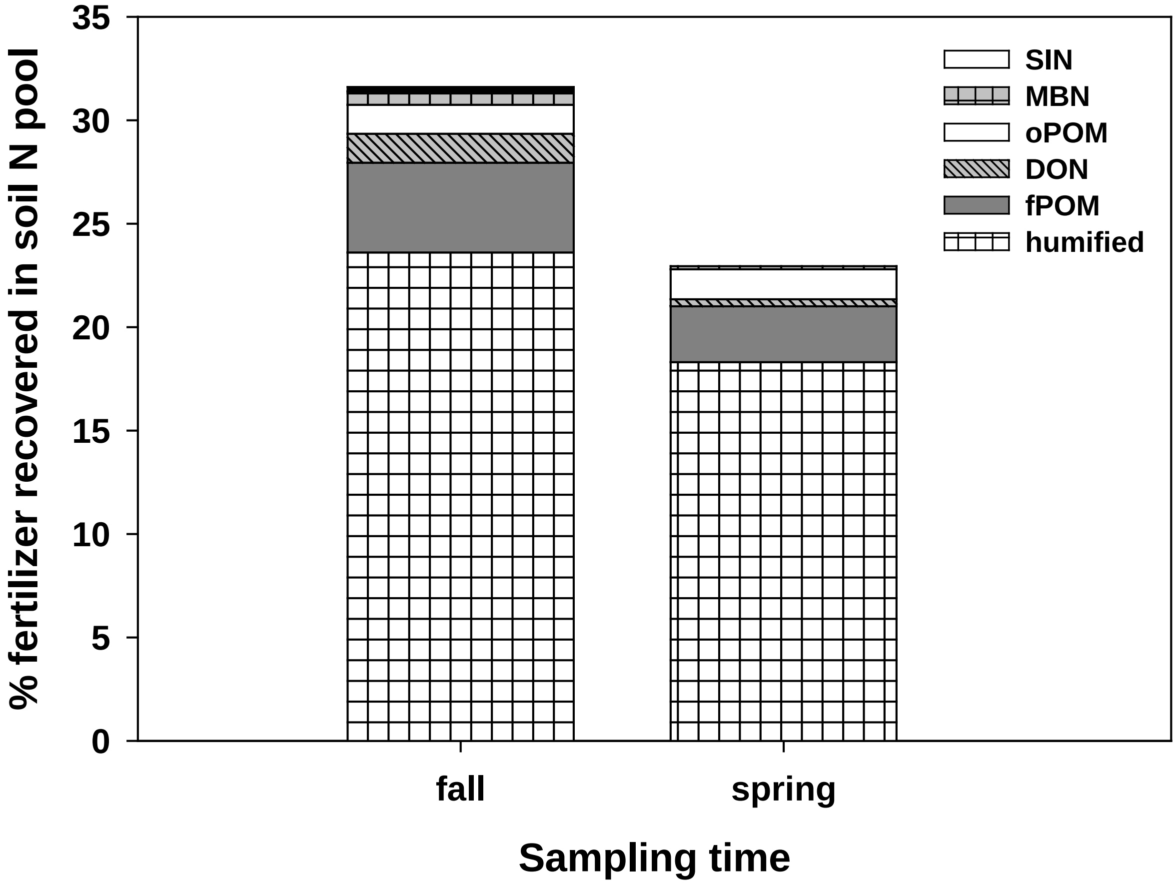


Figure 3.9. The % of added N fertilizer that was recovered in different soil N pools measured (to 1m) and the humified fraction (calculated by difference), at the fall and spring sampling times (average of BF and CC treatments).

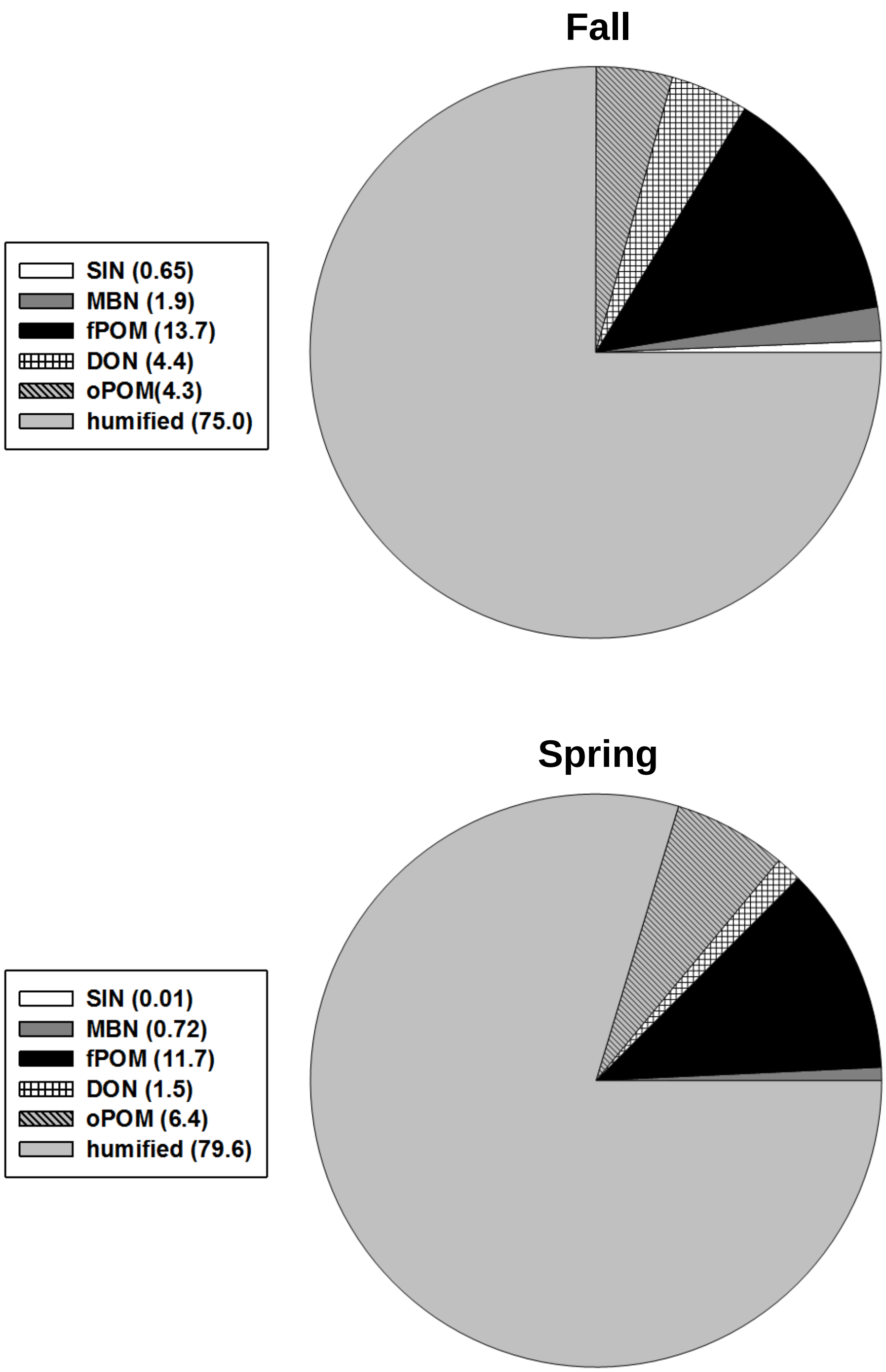


Figure 3.10. Distribution of ^{15}N-fertilizer recovered in the total soil N pool (i.e., as a percentage of the total fertilizer N recovered in the soil; in parentheses for each N pool in the legend) at the fall and spring sampling times (average of BF and CC treatments). In the fall 47.4 kg N ha^{-1} of fertilizer was recovered in soil to 1 m (i.e., 31.6% of added) and in the spring 34.5 kg N ha^{-1} was recovered (23.0% of added).

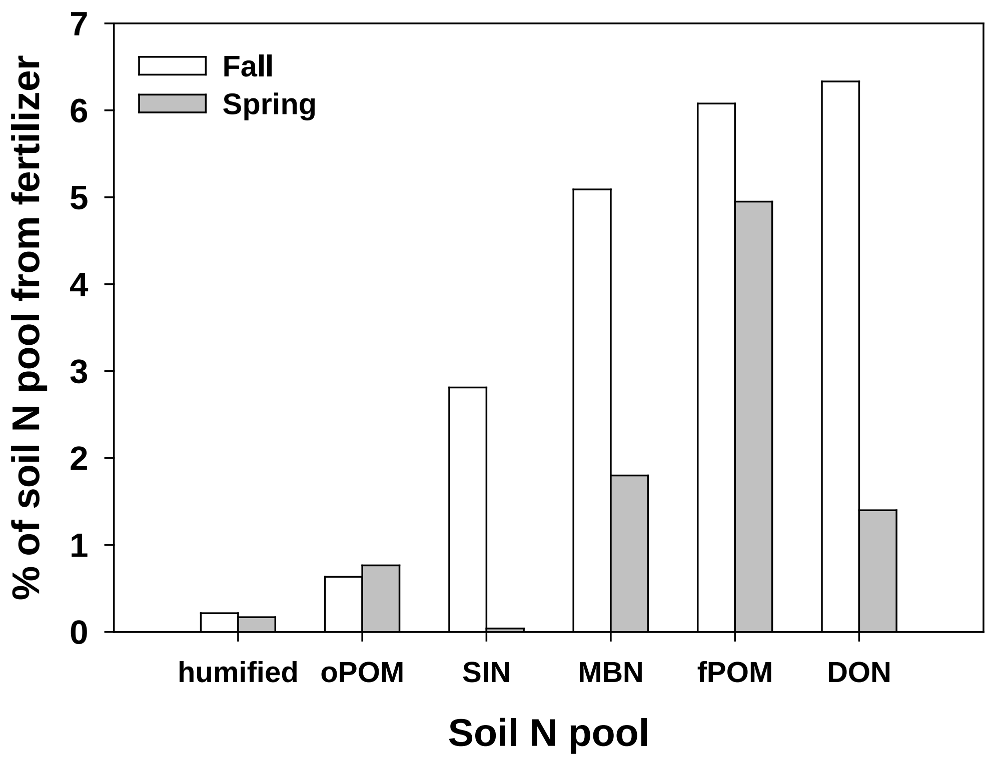


Figure 3.11. The fraction (%) of each soil pool that is from fertilizer for the different soil N pools measured, and the humified fraction by difference, at the fall and spring sampling times (average of BF and CC treatments).

Discussion

Recovery of ^{15}N in Corn

Fertilizer recovery in corn at harvest fell within the typical range for ^{15}N experiments in conventional corn-soybean agroecosystems, but was on the low end. For example, whole plant % recovery of ^{15}N ranged from 17-27% (Table 3.2), whereas Blackmer and Sanchez (1988) found a similar range of % recovery in the grain portion of the plant, and whole plant % recoveries ranging from 31-44% of added ^{15}N. The wet, cool summer in our experiment led to lower corn yields than usual (Table 3.1), which is likely the reason for slightly less ^{15}N recovery in corn. Despite slightly lower yields and % recovery of added fertilizer, the % of corn N that came from fertilizer (44%) in our study corresponds well with the mean of other corn ^{15}N studies (Gardner and Drinkwater, 2009). This suggests that the partitioning of SOM and fertilizer N sources in corn is relatively fixed and that corn typically relies on SOM for the majority of its N.

We also found that N fertilization stimulated decomposition of SOM (Figure 3.2B; i.e., the "priming effect") because corn in the two fertilized treatments took up more soil N than corn in the unfertilized control. Two potential mechanisms could account for the greater plant uptake of soil N: i) fertilization led to direct stimulation of microbial decomposition of SOM, and/or ii) fertilization indirectly increased SOM decomposition through greater crop growth and plant-stimulated decomposition via the microbial loop (Clarholm, 1985). Since the treatments received the same amount of fertilizer, it is likely that the second mechanism was more important in our study because there was significantly greater soil N uptake in the BF treatment with slightly higher corn biomass (Figure 3.2B). These results suggest that isotope methods are an important methodological tool because the difference method, which is also commonly used in studies of fertilizer and N cycling (e.g., Schindler and Knighton, 1999), does not distinguish between the different N sources in plants or account for the priming effect of N fertilization.

Corn yield and temporal dynamics of ^{15}N recovery in SOM pools

The majority of ^{15}N literature in agroecosystems focuses on crop uptake of inorganic fertilizer in a single growing season (Gardner and Drinkwater, 2009), which misses the most important window for N losses (McCracken et al. 1994). Here, we followed the fate of ^{15}N fertilizer over a full year, and into both crops and soil, which is an ecosystem-based perspective that permits calculation of ^{15}N mass balances. In addition, we followed the added ^{15}N into a range of heterogeneous soil N pools with differing turnover times and biochemical composition to improve understanding of soil N cycling in grain cropping systems. Although the unexpected artifact of variability in yields among treatments (where there should have been none because those treatments had not yet been imposed) was a problem for addressing our research questions about winter cover, we can take advantage of that variability to explain and understand other dynamics in N-fertilized corn fields. For example, corn biomass impacts the temporal cycling of N fertilizer. Corn grain yield and total biomass play a large role in terms of ^{15}N recovery during the growing season (Figure 3.3A) and that recovery then impacts the movement of fertilizer N from litter into SOM pools (Figure 3.3B; Tables 3.2 and 3.3). That is, in the absence of winter cover, the amount of N recovered in the corn in the fall drives total recovery in the system overall.

The treatment with greater biomass (BF) had greater corn litter returned to the field and greater recovery of fertilizer N in labile N pools such as DON, MBN, and fPOM (Tables 3.2 and 3.3) showing that this is the dominant pathway of ^{15}N recovery in this system. We also found some evidence for movement of N from the fPOM pool into the oPOM pool as decomposition occurred over the course of the year. In the fall the fPOM pool for both treatments had a C:N of 25 and the oPOM pool had a C:N of 20. In the spring the fPOM C:N was the same in the BF treatment and was 30 in the cover crop treatment, potentially reflecting recent inputs of rye C in the CC plots. In the spring, the oPOM C:N in the BF treatment increased to 30, perhaps indicating that corn litter from the fPOM pool was being decomposed and moving into the oPOM pool. At this point the corn litter would likely be dominated by root residue, since root-derived C persists longer in the soil and is more often incorporated into oPOM pools than shoot-derived C (Puget and Drinkwater, 2001).

These findings support our hypothesis that a greater proportion of the N in the labile SOM fractions with faster turnover times would be from newly added fertilizer; however, we found that the greatest proportion of added fertilizer N that was recovered in the soil was in the humified OM pool at both sampling dates. This agrees with the findings of other ^{15}N tracer studies in forest ecosystems (Nadelhoffer et al. 1999, Nadelhoffer et al. 2004). In an incubation study, Moran et al. (2005) found that the addition of a C-source promoted the incorporation of inorganic N fertilizer into humin. We similarly found that the treatment with slightly greater corn biomass (BF) had greater ^{15}N retention overall, and greater ^{15}N retention in the total soil N pool (Table 3.3). The added N is likely immobilized by microbial biomass to a greater extent in the presence of greater C (i.e., C and N cycles are more coupled), however, the biotic and abiotic mechanisms behind this observation are not well understood, and more research is needed to explain N assimilation in soils, particularly in the humified fraction.

Though the relationship between corn yield and ^{15}N recovery provides useful information about the mechanisms and pathways of ^{15}N incorporation in grain agroecosystems via biomass, the focus on increasing corn yields and crop uptake of added fertilizer alone is not sufficient. For one reason, studies that only measure crop uptake of added ^{15}N often have difficulty with interpreting results due to the complexity of the soil N cycle. That is, the biological exchange of labeled N with unlabeled soil N through microbes and through various labile SOM pools impacts the partitioning of ^{15}N between aboveground and belowground pools. Some researchers focused on measuring ^{15}N recovery in corn have suggested that this exchange of labeled and unlabeled N due to microbial turnover is a methodological problem with tracer studies that can lead to errors in calculating crop fertilizer use efficiency (e.g., Jenkinson et al. 1985; Schindler and Knighton, 1999). However, the movement and cycling of N through above- and belowground pools is, indeed, what happens to fertilizer when it is applied to fields—it enters the complexity of the N cycle and is subject to plant and microbial competition processes (Hodge et al. 2000). This highlights the importance of ecosystem-based research that considers the fate of N in crops *and* soil, rather than just crops. Results of tracer experiments are thus valid as long as the assumptions of isotope methods are met (Hauck and Bremner, 1976; Stark, 2000).

In this experiment the total % recovery of added ^{15}N in the spring was 45% in BF (67.5 kg N ha^{-1}) and 37% in CC (55.3 kg N ha^{-1}). This mass balance accounts for total recovery in rye and soil in the spring, and the amount of ^{15}N that was exported with corn grain N in the fall. Thus, overall, N losses during the year were large. On average, 38% of added N is unaccounted for in the ^{15}N literature in temperate grain cropping systems (Gardner and Drinkwater, 2009). Our losses were larger than this average (55-63%); however, the mean for other studies is from measurements taken at the end of one growing season, which does not account for the winter bare fallow period. In addition, the very wet summer and fall likely increased losses of soluble N through both increased leaching and decreased corn yields and N uptake. Therefore, conventional cropping systems that do not couple N inputs with C (e.g., through use of organic sources that need to mineralize before becoming available, or through increased temporal plant diversity) are inherently leaky and must be maintained in a N-saturated state to achieve high yields. Winter cover crops could potentially play a role in improving the N use efficiency of these systems (Tonitto et al. 2006).

Role of winter cover in retention and cycling of ^{15}N

We did not find support for our hypothesis that a winter rye cover crop would increase agroecosystem recovery of ^{15}N-labeled ammonium sulfate fertilizer compared to a winter bare fallow in a tile-drained Illinois Mollisol. The low recovery of labeled fertilizer in the cover crop is similar to findings reported in other ^{15}N cover crop studies, though the amount of total N and ^{15}N fertilizer in the cover crop in our study is smaller. The extremely rainy October (Figure 3.1) likely led to losses of most of the inorganic N forms prior to treatment establishment. We were therefore also unable to effectively test the mechanisms of ^{15}N cycling in the presence of a winter rye cover crop because weather conditions and resulting poor establishment of the cover crop meant that the rye growth missed the optimal window for ^{15}N recovery. The total soil inorganic N pool in this experiment (Tables 3.2 and 3.3) was very small compared to other ^{15}N studies (e.g., Kissel et al. 1977; Tran and Giroux, 1998), suggesting that losses from this pool had occurred in the fall.

Despite the poor cover crop growth (Figure 3.5) and low recovery of ^{15}N, the cover crop scavenged inorganic N in the CC treatment compared to the BF treatment (Figure 3.6). Thus, even in a year with limited cover crop growth, we found that the cover crop played a key N-scavenging role, likely reducing total N losses in the spring following snowmelt and soil warming. We found multiple lines of evidence supporting this mechanism— the total N uptake by rye biomass (23.7 kg N ha^{-1}) was comparable to the difference in the SIN pool between the

CC and BF treatments (11.1 kg N ha^{-1}), though plant N uptake provides a more integrated metric than the ephemeral soil NO_3^- and NH_4^+ pools. Further, within the CC treatment plots, there was a negative relationship between rye biomass and the soil NO_3^- pool (Figure 3.7). These findings support other literature on cover crops reducing N leaching (Tonitto et al. 2006), and highlight the potential ecosystem services they can provide in agroecosystems. In a recent cover cropping study, McSwiney and others (2010) also found decreased NO_3^- concentrations with a cereal rye cover crop, and reduced potential for NO_3^- leaching. The authors concluded that fertilizer N immobilization due to cover crop incorporation is a useful farm management tool that can be optimized to reduce N losses and maintain crop yields.

Similar to the few other ^{15}N studies with cover crops in corn-soybean systems, we found that a very small amount of fertilizer N was recovered in the rye biomass. In our experiment, the poor rye growth and late timing of rye growth is one explanation; however, the results also appear to support our hypothesis that the role of a cover crop is largely in recovering N derived from mineralization of SOM rather than newly added fertilizer N. This is also supported by our finding that the majority of fertilizer N in the soil was in the large fraction of 'humified' OM, rather than in the pools with faster turnover times that we measured in both fall and spring (Figures 3.9 and 3.10). Thus the cover crop conserves N by scavenging N released through mineralization as the soil warms in the spring, and also potentially by providing root C exudates to fuel microbial incorporation of fertilizer into SOM pools, rather than directly taking up that N. However, we weren't able to test the latter mechanism in this experiment. In the spring, the influence of the cover crop biomass on SOM was evident in the slightly larger total fPOM N pool size in the CC treatment (Table 3.3). Other studies have reported that more diverse cropping systems, such as organic farms, have larger POM pools than fertilizer-based cropping systems

(Marriott and Wander, 2006). We expect that use of cover crops would lead to increases in the size of POM pools and the capacity for the soil N cycle to supply crop N (Nissen and Wander, 2003).

Conclusion

Results from our study contribute to the literature documenting the fate of N in conventionally-managed, high-input agroecosystems, indicating that in a year in which grain yields were limited, overall recovery of added fertilizer was only 37-45%. This study also makes a contribution to a much smaller literature that follows the fate of ^{15}N into a range of labile and stable SOM pools that are important in soil N cycling. For example, in line with other ^{15}N literature (Gardner and Drinkwater, 2009), only 44% of the N in corn came from fertilizer at the end of the growing season. Thus, decomposition of SOM plays a large role in crop productivity and we found some evidence for corn-stimulated decomposition of SOM (Figure 3.2) in this study. Further, even in a high-input grain system where N inputs are not coupled with C inputs, we found evidence that C and N biogeochemical cycling is linked. The treatment with greater corn yields (BF) had slightly greater retention of ^{15}N overall, and significantly greater ^{15}N recovery in a subset of labile soil N pools that led to cascading differences in responses between the treatments post-growing season. The potential for managing agroecosystems to intentionally re-couple C and N inputs and cycling to improve NUE has been supported by meta-analyses (Gardner and Drinkwater, 2009; Tonitto et al. 2006), long-term cropping systems experiments (Drinkwater et al. 1998; Ross et al. 2008), and on-farm studies (Blesh and Drinkwater, *in prep.*), all of which have demonstrated that complex crop rotations with winter cover and use of legume N sources lead to reduced potential for N losses.

In this study, we aimed to test how the addition of a winter cover crop (i.e., a C-sink for fertilizer N) in a conventional grain cropping system would impact the retention and cycling of ^{15}N. Though the results of field experiments are more representative of reality than greenhouse experiments, research conducted in field conditions is subject to real weather conditions, and the year in which we conducted this experiment was not favorable for cover crop establishment. The weather conditions together with unexplained variability in corn biomass during the growing season were the major drivers of reduced absolute retention of fertilizer in the CC treatment in the spring. Thus, our questions about the mechanisms of cover crop contributions to N use efficiency remain open. The policy- and research-focus on maximizing corn yields has led to breeding for corn varieties with earlier planting dates, thereby making the introduction of cover crops into this type of agroecosystem difficult. The results of this study, and the challenges with cover crop establishment, may not, in fact, be unusual—this study may reflect the current challenges to planting and successfully establishing cover crops in a system where the window for cover cropping has been gradually reduced over time. It is potentially common that farmers plant cover crops too late to capture the pulse of N loss in the fall, if they plant them at all. Policy incentives that include an approximate date by which a cover crop must be planted could be one option for increasing the effectiveness of cover cropping. Understanding *how* winter annual cover crops impact agroecosystem N cycling, and developing cropping systems that permit the effective establishment and growth of cover crops, are important areas for future inquiry that would help reduce N pollution from agricultural ecosystems.

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