CHAPTER FOUR

A New GIS Nitrogen Trading Tool Concept for Conservation and Reduction of Reactive Nitrogen Losses to the Environment

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Abstract

Nitrogen (N) inputs to agricultural systems are important for their sustainability. However, when N inputs are unnecessarily high, the excess can contribute to greater agricultural N losses that impact air, surface water, and groundwater quality. It is paramount to reduce off-site transport of N by using sound management practices. These practices could potentially be integrated with water and air quality markets, and new tools will be necessary to calculate potential nitrogen savings available for trade. The USDA-NRCS and USDA-ARS Soil Plant Nutrient Research Unit developed a web-based and stand-alone Nitrogen Trading Tool (NTT) prototype. These prototypes have an easy-to-use interface where nitrogen management practices are selected for a given state and the NTT calculates the nitrogen trading potential compared to a given baseline. The stand-alone prototype can also be used to calculate potential savings in direct and indirect carbon sequestration equivalents from practices that reduce N losses. These tools are powerful, versatile, and can run with the USA soil databases from NRCS (SSURGO) and NRCS climate databases. The NTT uses the NLEAP model, which is accurate at the field level and has GIS capabilities. Results indicate that the NTT was able to evaluate management practices for Ohio, Colorado, and Virginia, and that it could be used to quickly conduct assessments of nitrogen savings that can potentially be traded for direct and indirect carbon sequestration equivalents in national and international water and air quality markets. These prototypes could facilitate determining ideal areas to implement management practices that will mitigate N losses in hot spots and provide benefits in trading.

1. Introduction

The use of nitrogen (N) inputs in agricultural systems has heavily influenced the sustainability and economical viability of agricultural systems worldwide. These N inputs help maximize yields, which is necessary to supply food to the ever-growing world population. However, when these N inputs are higher than necessary, the excessive N can contribute to greater agricultural N losses that impact air, surface water, and groundwater quality (Fig. 1). One of the reasons that excessive N can lead to increased losses is that it is a very mobile and dynamic nutrient. Fortunately, best management practices for N can be used to synchronize N inputs with crop N uptake sinks in a way that minimizes N losses to the environment.

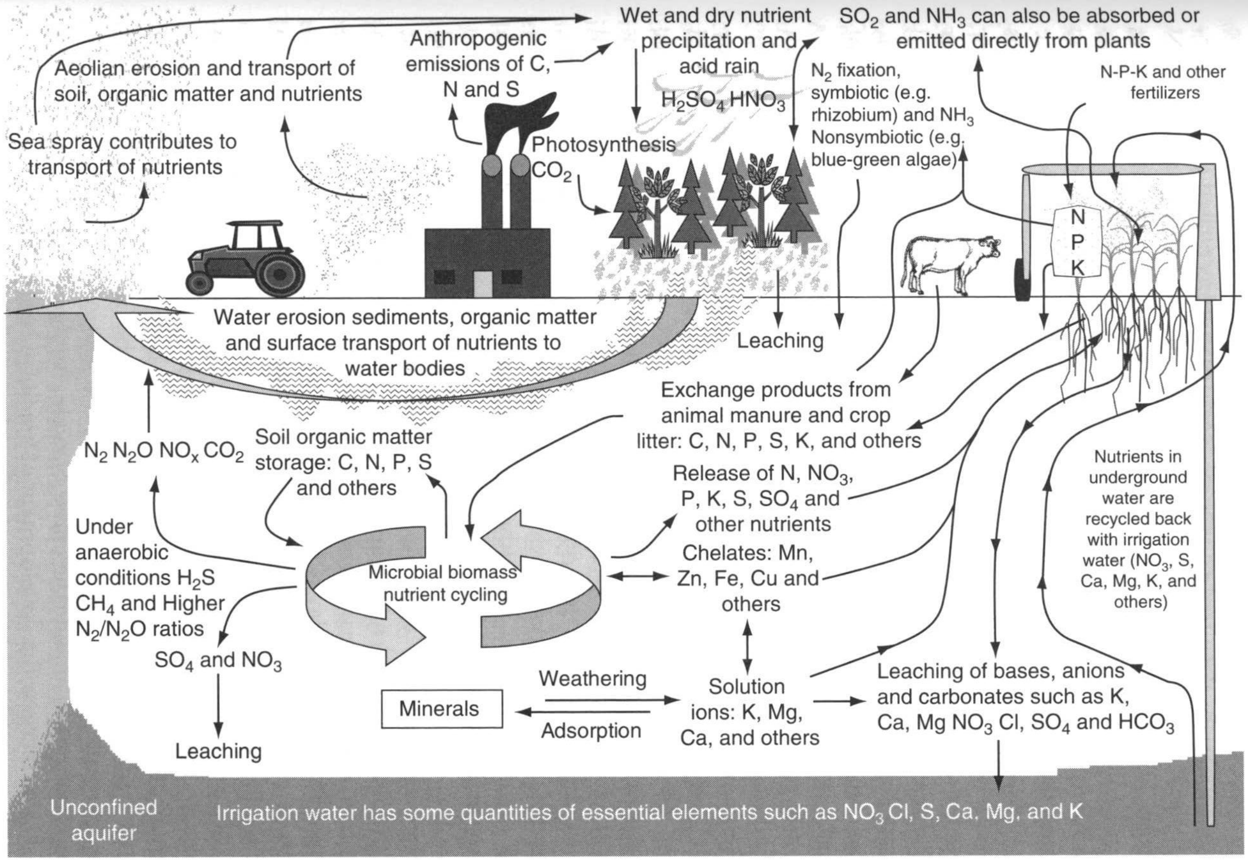

Figure 1 Nutrient cycles of essential elements for crop production, showing their fate and transport in the environment (from Delgado and Follett, 2002).

Most agricultural systems are naturally deficient in N, which makes N inputs necessary to maximize yields, crop quality, and economic returns required to sustain viable operations. This is especially true for intensive irrigated systems with higher average yields than nonirrigated systems, particularly during times when crops are growing faster and have greater N uptake. Nitrogen inputs to agricultural systems are very important for the sustainability of these systems. A key positive feature of N inputs is their contribution to crop yields and crop quality, which ensure higher economic returns for farmers. Another positive feature of N inputs is that they reduce the need to cultivate low-productivity agricultural land, allowing those areas to be left alone and allowing farmers to cultivate areas more suitable for agricultural production. Nitrogen inputs also contribute to higher water use efficiencies (kg mm^{-1} ha^{-1}), which are increasingly necessary for global sustainability as water resources in some regions become depleted. However, across any landscape system combination, any N application in excess of what is needed can increase the risk of negative effects on the environment.

It is paramount to reduce the off-site transport of N from fields with sound management practices. In order to continue the efforts to minimize agriculture's negative impacts on the environment, we need to continue developing and implementing best management practices for N at a field level. Even after N has left the boundaries of a field, there are other conservation efforts that can help identify areas of higher N transport (hot spots). Specifically, precision conservation techniques around fields and across water pathways and off-site management practices such as buffers, filter strips, riparian zones, sediment ponds, denitrification traps, irrigation and drainage ditches, and other management of natural areas within a watershed can help reduce reactive N transport across the landscape. For example, some researchers have proposed that we can even harvest N and reduce its transport across water bodies by using information about N dynamics to determine the best strategic placement of wetlands as a practice that can increase denitrification and removal of nitrates (NO_3-N) from surface waters (Hey, 2002; Hey *et al.*, 2005). We suggest that these nutrient management concepts and principles could potentially be used to reduce N transport in the environment.

We propose that the best approach to reduce off-site N transport is to work at a field level, starting with a good conservation and nutrient management plan that reduces excessive N inputs. We believe that the application of an N trading concept could help increase the implementation of best management practices for N at the field level and expand management considerations to include the entire N cycle. Applying an N trading concept could also increase the development of precision conservation at a watershed level that could include strategic placement and management of nutrient farming devices such as denitrification traps and better management of irrigation and drainage ditches and wetlands that reduce off-site N transport.

Improvement of N management, including the use of precision conservation practices across agricultural systems worldwide, will be critical to the sustainability of agriculture, maximization of yields, and the conservation of our biosphere during the twenty-first century. These practices will become even more important in this century if we are to reduce the continual increase in nitrous oxide (N_2O) emissions, which may contribute to global warming, and atmospheric reactive N deposition, which impacts the ecological balance in natural systems.

The increasing demand for biofuels presents another reason for conservation-focused N management. Since most of the agriculturally viable land in the world is already being used to produce food for the current population (Baligar *et al.*, 2001), the world population continues to increase, and biofuel cropping can compete for land area and water resources that are already being used for food production, and the sustainability and productivity of agricultural land are of utmost importance. Additionally, removal of crop residue may increase nitrate (NO_3-N) leaching and N_2O-N emissions (Delgado *et al.*, 2010) and erosion (Lal, 1999). Considering the continued reports about the possibility of global warming, climate change, extreme weather events (droughts and floods), depletion of important aquifers in some of the most productive regions in the world, desertification, a rise in sea level, and other ecological events that may impact food production, sustainable practices and maximum production in all agroecosystems will be necessary to ensure future worldwide food security (Eggleston *et al.*, 2006; Hatfield and Prueger, 2004; Houghton *et al.*, 1992; Hu *et al.*, 2005; Lal, 1995, 2000; Nearing *et al.*, 2004). Increasing the sustainability and yield per unit area will also relieve the pressure to cultivate marginal lands and forested areas, pressure that is otherwise likely to increase with population.

Several authors have reported on how excessive N applications that increase the potential for N cycle leaks can impact the quality of air, surface water, and groundwater (Follett and Walker, 1989; Follett *et al.*, 1991). For example, excessive N applications increase the potential for NO_3-N leaching losses, which can impact groundwater quality (Hallberg, 1989; Juergens-Gschwind, 1989). Anthropogenic N sources have been tied to losses of nitrogen that contribute to the Gulf of Mexico Hypoxia (Antweiler *et al.*, 1995; Goolsby *et al.*, 2001). Increased N inputs are also tied to increased emissions of trace gases such as N_2O, which increase the potential for global warming (Eggleston *et al.*, 2006; Houghton *et al.*, 1992; Mosier *et al.*, 1991). Other pathways that contribute to N losses include off-site surface transport (Bjorneberg, *et al.*, 2002) and ammonia (NH_3-N) volatilization (Peoples *et al.*, 1995), which impact water and air quality, respectively.

It has been shown that N management can improve the synchronization of N sources and sinks, but knowledge about how weather, the hydrologic cycle, irrigation, off-site factors, and cropping systems interact with the soil

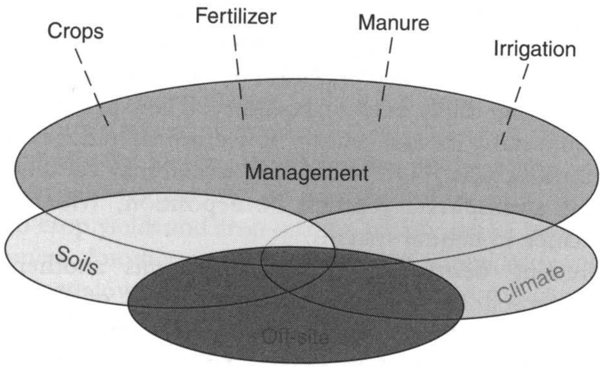

Figure 2 Essential components of NO_3-N Leaching Index (NLI) (from Shaffer and Delgado, 2002).

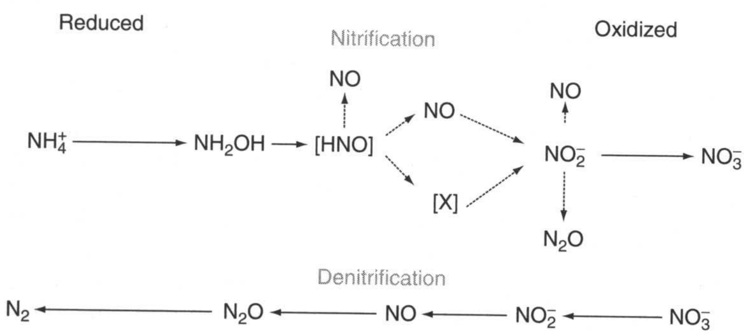

Figure 3 Diagram of nitrification and denitrification processes (from Mosier *et al.*, 2002).

N pools, N dynamics, and N fate and transport in a given landscape is invaluable (Delgado and Shaffer, 2008; Shaffer and Delgado, 2002) (Fig. 2). Basic management principles are often all that are necessary to minimize both NO_3-N leaching (Meisinger and Delgado, 2002) and N_2O emissions (Mosier *et al.*, 2002) (Figs. 1 and 3). Delgado and Lemunyon (2006) reported that it is important for nutrient managers to continually seek education regarding nutrient management to stay current in the newest advances in technique and technology. Knowledge of these advances is critical to ensure good, effective management decisions (Delgado and Lemunyon, 2006).

There are new tools that can be used to supply and integrate some of this information, which can help nutrient managers understand the potential for N loss savings resulting from the implementation of precision conservation management. New advances and practices such as controlled-release fertilizers, management zones, remote sensing, growing season *in situ* testing, cover crops, and limited irrigation are proven improvements to N management that result in significantly reduced N losses. Trading systems may become significant considerations during the management implementation process in the future. In this chapter, we present the concept of using a GIS Nitrogen Trading Tool (NTT) approach to assess N management and conservation practices to reduce reactive N losses to the environment. We propose that a GIS NTT based on computer models can help identify where the higher N losses are occurring across a field and how much savings in N may be achievable in a given field to be traded in water and air quality markets.

2. Understanding the Nitrogen Cycle with Respect to Nitrogen Management and Trading

Nitrogen management principles that can be used to increase nitrogen use efficiencies should be considered when evaluating the potential for increasing nitrogen trading. The NTT concept defined by Delgado *et al.* (2008c) assessed the differences in N losses between a new management scenario and a given baseline management practice. The NTT can conduct quick analysis about N management for the new scenario and baseline scenario using nitrogen and water budgets.

Since implementation of a new N management practice for 1 year can impact soil nitrogen pools and increase the release of nitrogen long after its initial application, the differences to the baseline are evaluated over a long time (24 years). This long-term evaluation integrates any changes to nitrogen pools or N sequestration (Al-Sheikh *et al.*, 2005) that could affect N dynamics. This could help ensure that the implementation of today's practices and the potential for trading will not create negative effects 5 or 10 years later due to changes in nitrogen dynamics. Because nitrogen management will affect the N pools and dynamics, it is important to use a mass balance for N and water budgets to track inputs and outputs over the long term.

It is important that the long-term evaluations take into account the interaction of management practices and field characteristics that consider the soil–crop–hydrologic cycle, which is site specific. To take advantage of best management practices that reduce N losses to the environment, we need to understand how the principles for nitrogen management can be used to reduce N losses and N transport to water bodies (Meisinger and Delgado, 2002; Randall *et al.*, 2008) and/or to the atmosphere (Mosier *et al.*, 2002). Additionally, an NTT that uses a mass balance analysis for nitrogen and water also helps to avoid simultaneously accounting for reductions in N inputs and losses.

Nitrogen use efficiencies have been reported to be around 50% in general and as low as 33% for cereals (Baligar *et al.*, 2001; Raun and Johnson, 1999). Baligar *et al.* (2001) discussed several different definitions of nutrient use efficiencies, including the nutrient use efficiency ratio (Gerloff and Gabelman, 1983), physiological efficiency, agronomic efficiency, agrophysiological efficiency, and apparent recovery efficiency. Delgado (1998) and Delgado *et al.* (2001a) assessed the effects of best management practice implementation on system N use efficiency with a modeling approach. This modeling approach considered an N mass balance and how best management practices for N increased the N use efficiency, reduced NO_3-N leaching losses, and mined NO_3-N from underground waters. Evaluations of multiple cropping systems showed that the deeper-rooted crops acted like vertical filter strips, recovering NO_3-N from groundwater, as well as reducing NO_3-N leaching (Delgado, 1998, 2001; Delgado *et al.*, 2001a). Delgado *et al.* (2008c) proposed that a similar mass balance approach should be used to quantify the potential for savings in nitrogen that can be traded in water and air quality markets assuming the implementation of a determined set of management practices.

This new nutrient trading concept may provide an additional factor for consideration by managers deciding what practices to implement to increase N use efficiencies. Several other researchers have reported on the potential to use environmental quality market credits to account for reductions of agricultural N losses and prevention of their transport into water bodies (Glebe, 2006; Greenhalch and Sauer, 2003; Hey, 2002; Hey *et al.*, 2005; Ribaudo *et al.*, 2005). However, we need to be realistic and consider that the dynamics of the N cycle make the quantification of these reductions in N losses difficult, especially when one considers interactions with the temporally and spatially variable hydrologic cycle, weather, soils, management, crop rotations, and other uncontrollable and isolated factors (such as thunderstorms), which may increase leaching and/or denitrification (Delgado, 2002). Delgado *et al.* (2008c) and Gross *et al.* (2008) described the potential use of quick, new NTTs to help quantify the effect of conservation practices and N management on reactive N losses to the environment.

The new concept of the NTT was defined within the context of the N cycle and considers an N mass balance approach for the cropping systems (Delgado *et al.*, 2008c). The NTT difference in reactive N losses (NTT-DNL_{reac}) draws comparisons between a baseline and new management scenarios. A positive NTT-DNL_{reac} means that a new N management practice increases the savings in reactive N, while a negative number means that there are no savings in reactive N. In other words, a positive number means that there is potential to trade these savings, while a negative number means that there is no potential for trade. The NTT-DNL_{reac} can be thought of as a bank account balance. A positive number means that there is N in the bank for trade and a negative number means that there is no N in the bank to trade. The new GIS concept that we are presenting in this chapter can be applied across the field and considers spatial and temporal variability.

The new N trading concept, a stand-alone NTT, and a new Internet prototype of an NTT were developed by the Natural Resources Conservation Service (NRCS), in cooperation with the Agriculture Research Service Soil Plant Nutrient Research Unit (ARS-SPNR) (Delgado *et al.*, 2008c; Gross *et al.*, 2008) (Figs. 4 and 5). Both the web-based version and stand-alone prototype allow users of this new technology to quickly determine how many potential N credits their farming operations can generate.

The new Internet and stand-alone NTT are the only tools with the level of rigor to allow producers to calculate potential N credits for air and water quality markets as a function of conservation measure implementation. Environmental aggregators, brokers, and water quality traders may also use these tools (Delgado *et al.*, 2008c; EPA-WQTN, 2007). The development of the N trading concept and the NTT is part of the national agreement between the USDA-NRCS and the EPA Office of Water to participate in potential water-quality trading programs (EPA-WQTN, 2007). We suggest that such a tool could be used for air quality markets and for direct and indirect carbon sequestration equivalent markets. Further, we propose in this chapter that the new NTT-GIS can be used to quickly

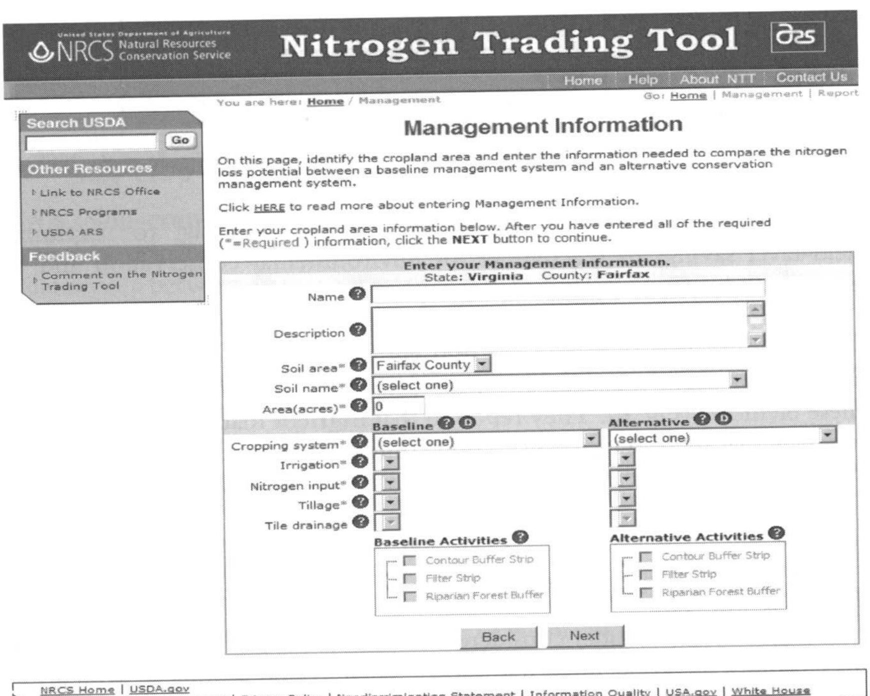

Figure 4 Nitrogen Trading Tool: the web-based prototype (from Gross *et al.*, 2008).

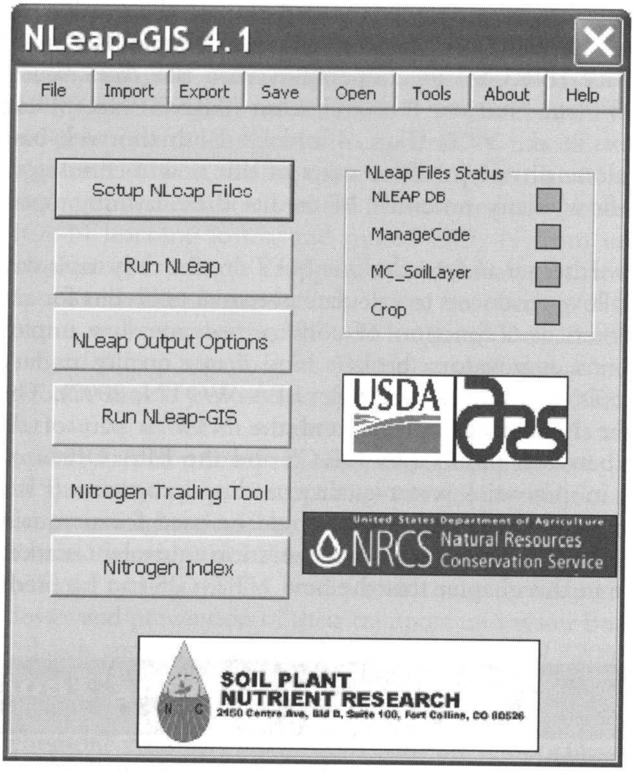

Figure 5 A stand-alone version of the NTT prototype (from Delgado *et al.*, 2008c).

identify the scenario that shows the greatest potential to maximize field-level savings in reactive N for environmental conservation and to earn N credits for trade.

Delgado and Follett (2002) reported that carbon management should also be a fundamental part of any nutrient management plan, integrating N and carbon input management with data about existing soil contents of these elements (Fig. 6). They reported that nutrient managers who manage in a way that increases the soil carbon content need to adjust for the greater N cycling and soil N mineralization potential by reducing N inputs (Fig. 6). In other words, we need to account for management practices that will increase soil organic matter (SOM) and N cycling by adjusting future N recommendations according to higher N mineralization rates (Delgado and Follett, 2002) (Fig. 6).

Delgado *et al.* (2008c) considered the complexity of the N cycle when they proposed a conceptual framework for nitrogen trading in which the effect of management practices is assessed using a computer model simulating a long period of time with an N mass balance approach. To avoid double accounting for N inputs, the N mass balance approach proposed by

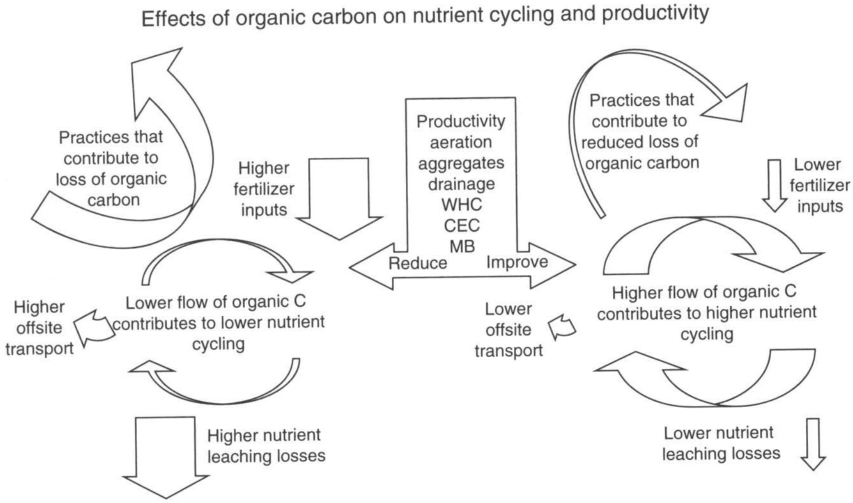

Figure 6 Potential organic C contribution to nitrogen cycling (+ organic C; + N cycling; − N fertilizer); use efficiency (+ organic C; + N use efficiency); nitrogen leaching (+ organic C; − N leaching); and nitrogen losses, under best management practices (from Delgado and Follett, 2002).

Delgado *et al.* (2008c) accounts for all N inputs and N outputs, as well as N transformations (e.g., N releases, sequestration, mineralization, etc.).

2.1. Understanding the relationships between the soil–crop–hydrologic cycle and nitrogen trading

2.1.1. Soil–crop–hydrologic cycle

One key principle for maximizing nitrogen trading is to understand the relationship between N management practices and the soil–crop–hydrologic cycle for a given region. This understanding could inform management decisions, and therefore help to avoid excess N applications while maintaining high crop yields, and help to increase the synchronization of applied N with crop N uptake sinks. Site-specific soil textures, hydrological properties, and crop water use all affect the soil water content and aeration and alter N dynamics (e.g., mineralization rates) and pathways for N losses (e.g., denitrification and nitrate leaching) (Meisinger and Delgado, 2002).

One example of the relationships between the hydrologic and N cycles is that high precipitation or irrigations can create water-logged conditions that favor potential losses of N due to denitrification (Meisinger and Randall, 1991). Other events that may contribute to water-logged conditions include the seasonal increase in water table. Meisinger and Randall (1991) summarized these relationships and reported that the denitrification potential for a well-drained soil with 1% SOM content on a semidry system will be about

3%. For the same case scenario under a humid or irrigated system, the denitrification potential increases by three times to about 9%. If the percentage of SOM is higher or if manure is applied, the denitrification potential will be higher (Meisinger and Randall, 1991). These denitrification losses are also driven by the lack of oxygen in the soil (Fig. 3).

Williams and Kissel (1991) reported on the interaction of soil hydrology and nitrogen losses. They reported that the threshold precipitation is less than 406 mm for dryland systems where NO_3-N leaching is zero or minimal. Evans *et al.* (1994) and Westfall *et al.* (1996) reported a similar relationship between the hydrologic cycle and NO_3-N leaching. However, when there are high precipitation events, NO_3-N leaching is much larger in soils with sandier and coarser texture that have hydrological properties that are conducive to a faster movement of water out of the root zone (Delgado *et al.*, 2001a; Follett and Walker, 1989). Williams and Kissel (1991) developed an index that incorporated these relationships between soil hydrological properties, weather, and water leaching.

Another pathway that contributes to N losses and is closely related to the hydrologic cycle is surface runoff. Surface transport caused by irrigation and/or precipitation is one means by which soil particles, SOM, organic N, and other N that may be bound to clay particles or dissolve in water can be transferred off-site and lost from the system. By understanding this property of the soil–crop–hydrologic cycle, nutrient managers could anticipate when periods of higher denitrification, leaching and/or erosion potential may occur for a given landscape crop combination, and implement conservation management practices that would reduce N losses.

Results from Williams and Kissel (1991) were adapted and presented in Figs. 7–9 using the same soil nitrate N concentrations across Ames, Iowa, Brookings, North Dakota, and Caldwell County, Kentucky. The adapted data from Williams and Kissel (1991) for the four major hydrologic groups in a high precipitation site such as Ames, Iowa, show that the potential for leaching is much higher in Ames than in a dryland region site such as Brookings, North Dakota (Figs. 7 and 8). These adapted data from Williams and Kissel (1991) are in agreement with the NTT results from Delgado *et al.* (2008c). They show that practices that reduce nitrate leaching will be advantageous for trading nitrogen on coarse texture sandier systems that more readily leach higher quantities of nitrate, especially under areas with higher precipitation or irrigation.

Seasonal timing of precipitation is also important to consider when managing nitrogen. For example, early or winter precipitation will help lead to increased leaching potential of available NO_3-N if there are no crops growing that can use water or uptake nitrogen, assuming there is nitrate available to leach in the soil profile. Figure 9 shows that the NO_3-N leaching losses were minimal when the precipitation was mainly occurring during the crop growing season (15 kg NO_3-N ha^{-1} y^{-1}). The same amount of precipitation, with a higher proportion occurring before planting,

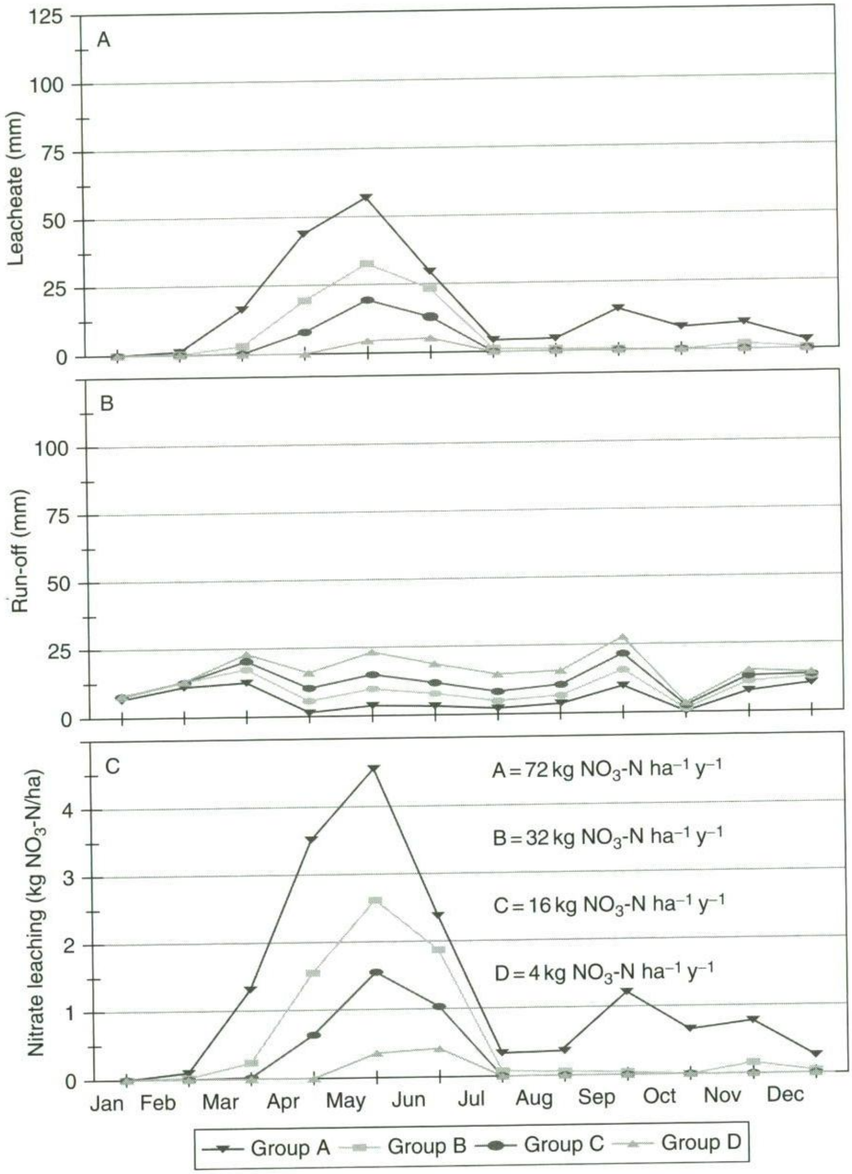

Figure 7 Leaching (A) and run-off (B) water volume (A and B adapted from Williams and Kissel, 1991). Estimated NO_3-N leaching (C) assuming same NO_3-N l^{-1} leachate concentrations for an irrigated corn grown in the main hydrologic soil groups (A, B, C, and D) under a rain-fed humid climate (Ames, Iowa) (from Delgado, 2004).

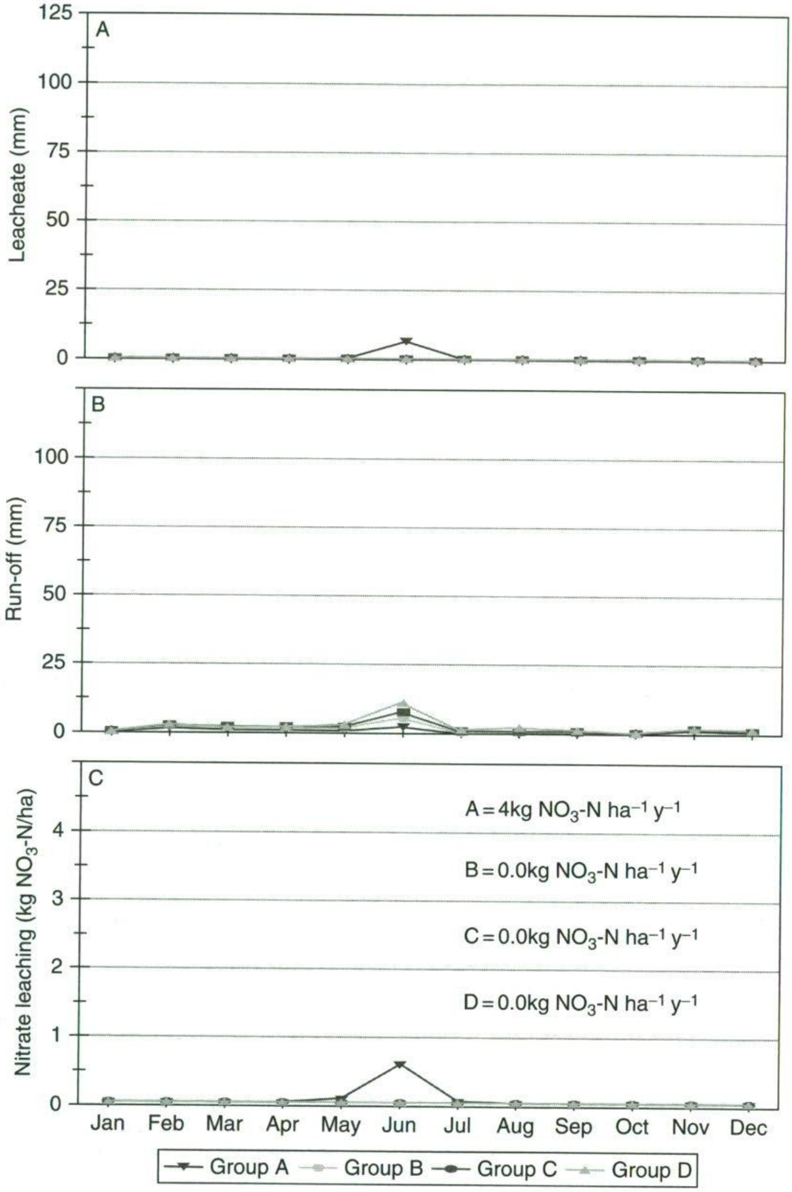

Figure 8 Leaching (A) and run-off (B) water volume (A and B adapted from Williams and Kissel, 1991). Estimated NO_3-N leaching (C) assuming same mg NO_3-N l^{-1} leachate concentrations for an irrigated corn grown in the main hydrologic soil groups (A, B, C, and D) under a rain-fed dry climate (Brookings, North Dakota) (from Delgado, 2004).

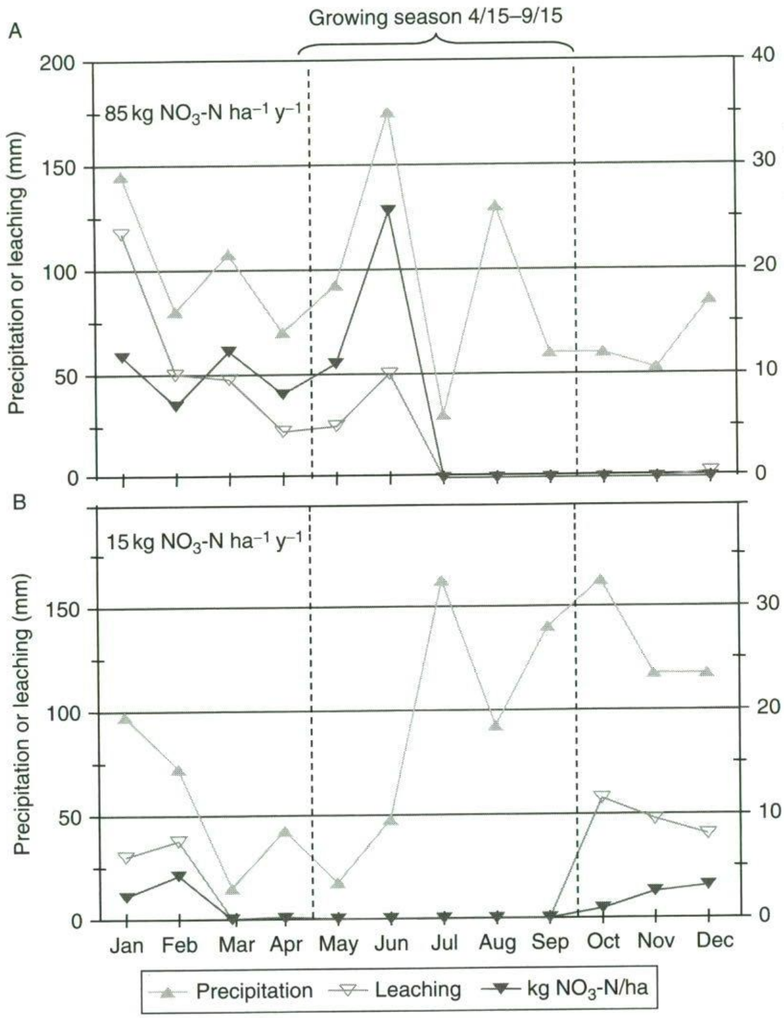

Figure 9 Effect of high (A) and low (B) in-season precipitation during corn growing season at Caldwell County, Kentucky (adapted from Williams and Kissel, 1991).

significantly increased the leaching potential by about six times if there was nitrate available to leach (85 kg NO_3-N ha^{-1} y^{-1}) (Fig. 9). Reducing the available NO_3-N to leach can reduce soil susceptibility to leaching during the winter months (Meisinger and Delgado, 2002).

2.1.2. Limited irrigation

Limited irrigation can improve water use efficiency while maintaining yields and a viable cropping system (Hu *et al.*, 2005). Delgado *et al.* (2007) reported that cover crops with limited irrigation can save water, reduce nitrate leaching, and even increase yields of subsequent potato crops. Incorporating a viable summer cover crop that can be grown with half of the water requirement of a traditional potato or winter wheat crop can save a significant amount of water. Figure 10 and Table 1 show the positive effects of using cover crops for hay and/or green manure across a region of south central Colorado. Limited irrigation has potential as a management tool to maintain viable cropping systems while increasing the potential for trading N and carbon sequestration equivalents (Delgado *et al.*, 2008b) (Fig. 11, Table 1).

2.2. Inputs

The average N use efficiencies are reported to be about 50% and as low as 33% for grains (Baligar *et al.*, 2001; Raun and Johnson, 1999); however, these N use efficiencies can be around 30% for irrigated shallow-rooted crops grown on sandy coarse soils (Delgado, 2001; Delgado *et al.*, 2001a,b) and lower than 30% when excessive N (750 to 1900 kg N ha^{-1}) is applied (Zhu and Chen, 2002; Zhang *et al.*, 1996). Nonetheless, for most agricultural systems N inputs are needed to maintain agricultural production, maximize yields, and quality and to supply the N that is removed with crop harvesting. Organic, inorganic, and biological (N fixation) sources can be used as N inputs for agricultural systems and may be applied using many different techniques.

2.2.1. Amount of N inputs

Management of N inputs can be done through a mass balance approach, in which all N sinks are considered along with crop uptake. Any N applied in excess of the crop uptake will increase the N available for leaching and the overall potential for N losses (Fig. 12). In order to increase the N use efficiency for the applied N, all N sources already present and available for uptake should be subtracted from the needed N. Examples of N sources to subtract include: residual soil NO_3-N that is available within the root zone or at least for the top surface foot, N that will be released from mineralization of SOM during the growing season, background NO_3-N applied with irrigation water while plant N uptake is active, and N that will be mineralized from the previous crop residue. Factoring the residue N from the previous crop is particularly important if the crop residue is from a leguminous crop that was incorporated into the soil or from a cover crop or vegetable crop with low carbon to nitrogen ratios.

A nutrient manager can calculate the needed N using an efficiency factor that accounts for management practices. Some states and regions have developed N uptake formulae that include efficiency factors to supplement data about N sink from crops. For example, in Colorado, the calculation of the appropriate amount of N fertilizer application to corn is based

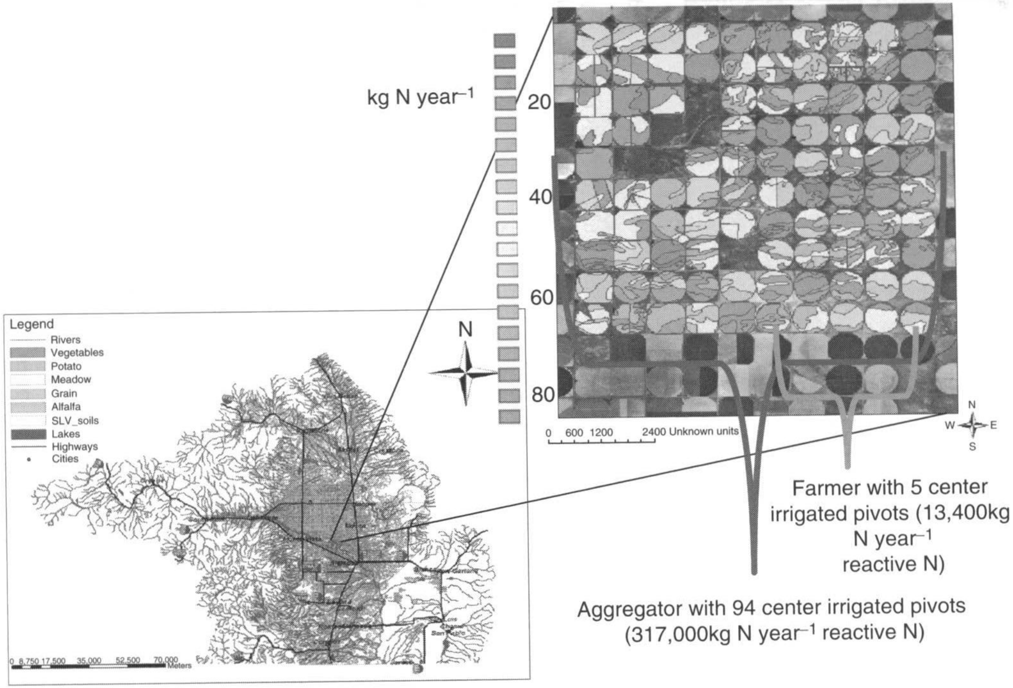

Figure 10 A stand-alone NTT-GIS prototype can be used to quickly evaluate the effects of management practices on total reactive N losses and the resultant potential to trade across regions (hypothetical example; adapted from Delgado *et al.*, 2008b).

Table 1 Assessment of potential reductions in nitrous oxide emissions (N_2O-N), in reactive N losses, and carbon sequestration equivalents

State	BMP[a]	N_2O credits[b] (kg N)	Total N credits[b] (kg N)	Carbon sequestration equivalents: Direct[c] (kg C)	Indirect[d] (kg C)	Total (kg C)
Virginia	Add legume	240	700	32,000	500	32,500
Virginia	Improved NM	70	900	9200	800	10,000
Ohio	Improved MM-Inc	350	7600	46,900	8800	55,700
Ohio	Improved MM-Spring	430	6200	56,500	6800	63,300
Colorado	Add SCC-LI	100	5300	13,700	5100	18,800
Colorado	Improved NM-CC	80	5000	10,200	5000	15,200

[a] NM, nutrient management; MM-Inc, manure management incorporated only spring application; MM-Spring, manure management only spring application; SCC-LI, summer cover crop with limited irrigation; CC, cover crop.

[b] The baseline for Ohio was a manure application in spring before corn planting (249 kg N ha^{-1}) and fall after soybean harvesting (also 249 kg N ha^{-1}). The baseline for Virginia was a continued conventional corn–corn rotation at 224 kg N ha^{-1}. The baseline for Colorado was a continued potato–potato rotation at 269 kg N ha^{-1}. The soils type across the 100 ha were loam, loamy fine sand and loam for Ohio, Virginia and Colorado, respectively.

[c] Direct carbon sequestration equivalents ($\Delta DCO_2 - C_{seN_2O}$) were calculated for 100 ha by using the equation: $\Delta DCO_2 - C_{seN_2O} = \Delta N_2O - N \times 310 \times 0.2727 \times 1.571$.

[d] Indirect carbon sequestration equivalents ($\Delta ICO_2 - C_{seN_2O}$) were calculated for 100 ha by using the equation: $\Delta ICO_2 - C_{seN_2O} = [((\Delta NO_3 - N + \Delta N_{st} - N + \Delta N_{er}) \times 0.0075 \times 310 \times 1.571) + (\Delta NH_3 - N \times 0.01 \times 310 \times 1.571)] \times 0.2727$.

The estimated potential for direct, indirect, and total carbon sequestration equivalents calculated using the nitrogen trading tool are also presented here for different best management practices (BMP).

on the Mortvedt *et al.* (1996) algorithm for N fertilizer applications: [N rate = 35 + (1.2 × EY) − (8 × soil ppm NO_3-N) − (0.14 × EY × OM) − (other N credits)], where EY is expected yield and OM is organic matter. The basic principle is to apply the correct amount of N, so as to avoid the excessive application of N. On average, using an N budget approach, whether by considering the N sinks and sources or using already calibrated formulas such as those developed by Mortvedt *et al.* (1996), will help increase the N use efficiencies by prescribing N inputs more closely aligned with the N needs of the system.

2.2.2. Types of N inputs

Several sources of inorganic N fertilizer are available. Among the most important are ammonia (NH_3), nitrogen solutions (combinations of ammonium nitrate (NH_4NO_3), urea, and water), ammonium nitrate, urea,

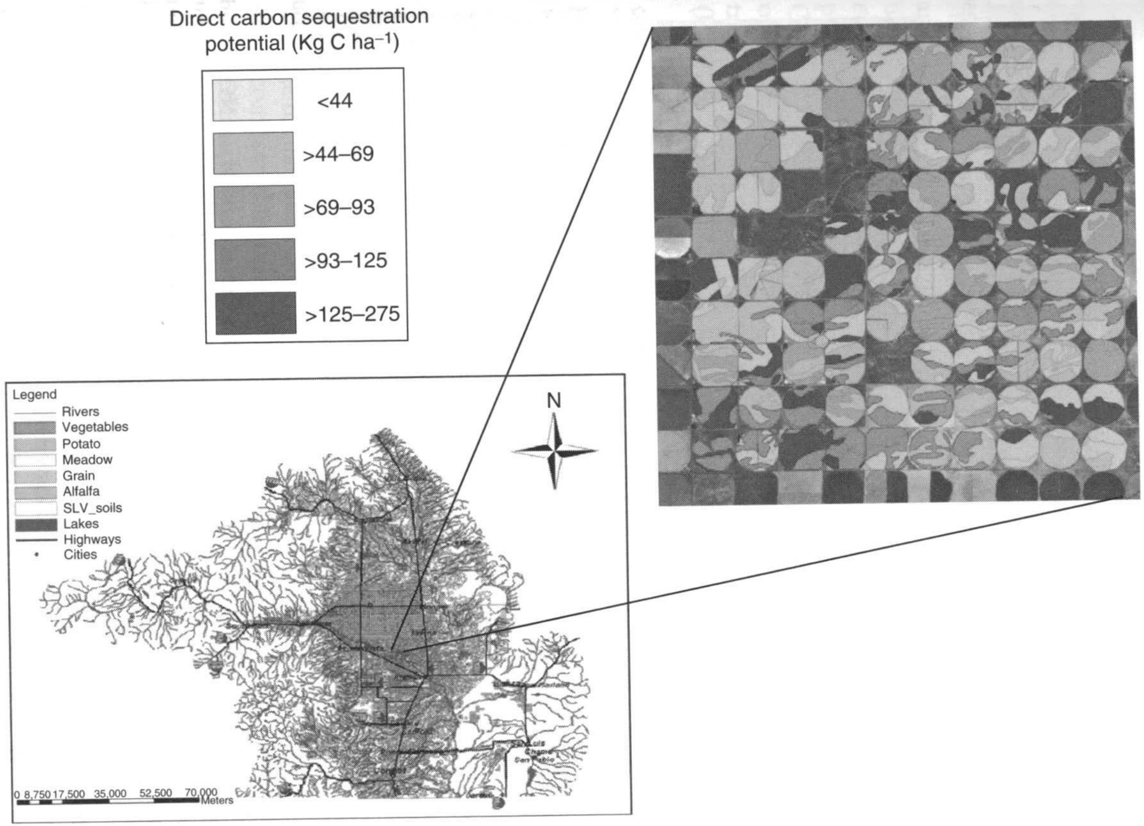

Figure 11 When released, the stand-alone NTT-GIS will be able to quickly evaluate the effects of management practices on direct carbon sequestration equivalents (kg C ha^{-1}).

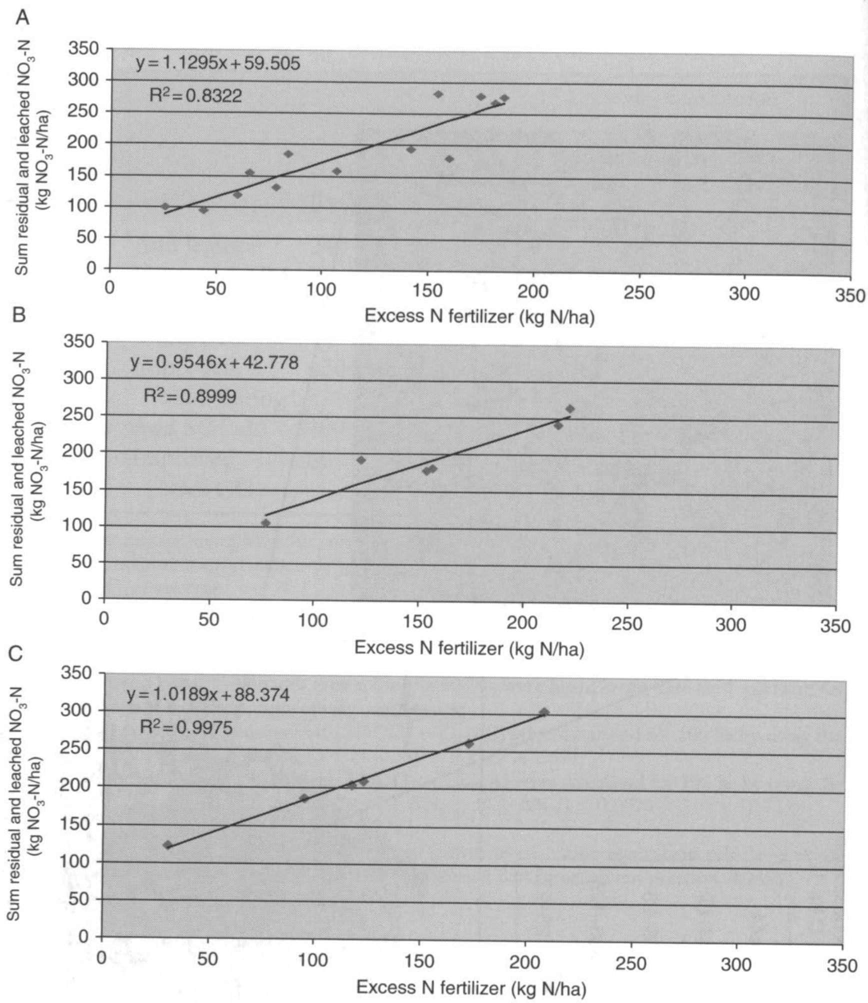

Figure 12 Correlation between excess N fertilizer and sum of residual soil NO_3-N and NO_3-N leached for low (A), medium (B), and high (C) productivity management zones. Residual and leached fertilizer was simulated with NLEAP (data adapted from Delgado and Bausch, 2005; Delgado *et al.*, 2005). Excess N use fertilizer was defined as: N fertilizer − (N uptake by crop − N uptake by control or zero fertilizer).

ammonium sulfate, and several other sources, such as ammonium phosphates and calcium nitrate (Boswell *et al.*, 1985). The source of N is important to consider when managing N, and careful choice of source can be used to increase the efficiency of a given system. For example, NO_3-N sources should not be applied to systems that will be submerged in water, such as rice fields, due to the high potential for N losses via denitrification. In a submerged system an NH_4-N source should be used instead. The NO_3-N sources can also increase the leaching potential on sandy irrigated soils; therefore, application of urea or NH_4-N sources, rather than NO_3-N application, can likely increase the nitrogen use efficiency and reduce the NO_3-N leaching potential for these systems.

Other N sources, such as controlled-release fertilizers, can be used to correlate the timing of N release with times of greater N uptake (Shoji and Gandeza, 1992). Controlled-release fertilizers contain the N source inside a capsule and release the N slowly to correspond better with periods of crop N uptake, thereby reducing the time that the N is susceptible to losses (Amans and Slangen, 1994; Mikkelsen *et al.*, 1994; Rauch and Murakami, 1994; Shoji and Gandeza, 1992; Shoji and Kanno, 1994; Wang and Alva, 1996). Several field studies have shown that, when using controlled-release fertilizer, nutrient managers can apply 50% of the traditional amount of fertilizer and still produce the same yields as with traditional fertilizer practices (Shoji and Gandeza, 1992; Shoji *et al.*, 2001). In other words, the fertilizer use efficiency of the controlled release fertilizer is much higher than that achieved using traditional fertilizer practices, helping to reduce agricultural N_2O emissions (Delgado and Mosier, 1996; Shoji and Gandeza, 1992, Shoji *et al.*, 2001).

Nitrification inhibitors (NI) can help increase N use efficiencies by slowing down the nitrification of NH_4-N to NO_3-N (Freney *et al.*, 1992; Yadvinder-Singh *et al.*, 1994). The NH_4-N is less susceptible to leaching, binds more to the clay particles, and is not affected by denitrification. Nitrification inhibitors also reduce the emissions of N_2O (Bronson and Mosier, 1993; Delgado and Mosier, 1996) and have been reported to reduce NO_3-N leaching (Owens, 1987; Timmons, 1984).

Organic N sources such as manure can also be used to provide N to agricultural crops. Significant amounts of manure N can be cycled to the subsequent crops (Eghball *et al.*, 2002). Eghball *et al.* (2002) reported that composted manure can cycle 18% of its N content during the first year, while cattle feedlot manure can cycle 30% of the N content. They reported that the total N available from feedlot manure is double the total N available from composted manure (Davis *et al.*, 2002; Eghball *et al.*, 2002).

Kirchmann and Bergstrom (2001) reported that N management is more important than N source in terms of controlling NO_3-N leaching losses when organic farming practices are compared to traditional farming practices. In either case, overapplication of N will contribute to increased NO_3-N leaching problems. They concluded that reduction in NO_3-N leaching was not as much a question of organic versus conventional farming as it was a question of adequate management practices. It is very important to practice effective N management with manure applications to avoid environmental degradation that can result from excessive application.

However, Delgado *et al.* (2010) reanalyzed unique ^{15}N crop residue exchange studies that used the Delgado *et al.* (2004) method and reported that N losses from organic crop residue are much lower (about 13%) than N losses from inorganic N fertilizer (about 31); these results conflict with the Kirchmann and Bergstrom (2001) study. Delgado *et al.* (2010) also conducted DAYCENT simulation analysis to evaluate the N losses from inorganic N fertilizer versus crop residue, and they found that the NO_3-N leaching losses and N_2O emissions were much lower from crop residues than from inorganic N fertilizer inputs.

2.2.3. Method and time of N inputs

The method by which N is applied, whether or not the N is applied in split (multiple) applications, the equipment used for application, and the location of application are important management factors that can be manipulated to increase N use efficiencies. It is important that we closely match the N inputs with N sinks (Meisinger and Delgado, 2002). The time of N application can be adjusted in order to reduce the time that the N is susceptible to losses if the periods of N availability are synchronized with the periods of more active rooting. For example, ^{15}N isotopic studies show that spring N applications are used more efficiently than fall applications (Delgado *et al.*, 1996). The spring ^{15}N isotopic fertilization recoveries in plants and soil were 60 and 71% for urea and NH_4NO_3, respectively, in contrast to the 42 and 57% recoveries from fall applications (Delgado *et al.*, 1996).

Several scientists have reported on the benefits of splitting N applications into preplant, side-dress, and fertigations in order to match greatest N availability with the periods of greatest N sinks (Gunasena and Harris, 1968; Oberle and Keeney, 1990; Russelle *et al.*, 1981; Sowers *et al.*, 1994; Stanford and Legg, 1984; Westermann and Kleinkopf, 1985). Split N applications that reduce the amount of total N applied and increase the number of N applications will improve N use efficiency and crop yield while reducing the potential for N losses (Alva and Paramasivam, 1998).

Good water management practices are important to increase N use efficiencies and reduce NO_3-N leaching losses to the environment (Meisinger and Delgado, 2002). There are best management practices that can help minimize NO_3-N leaching losses (Alva and Paramasivam, 1998; Hergert, 1986; Smika *et al.*, 1977; Thompson and Doerge, 1996a,b; Westermann *et al.*, 1988). Management systems under sprinkler irrigation that use fertigations can contribute to higher N use efficiencies, especially for shallower-rooted cropping systems and vegetables that are grown in sandier coarse textured soils with a lower capacity to hold water (Westermann *et al.*, 1988). For coarser soils, a high number of fertigations (5–8) help increase N use efficiencies (Doerge *et al.*, 1991). The application of N below the surface can increase N use efficiencies compared to broadcast methods, especially when NH_3-N volatilization is reduced (Meisinger and Randall, 1991; Peoples *et al.*, 1995).

2.2.4. Advanced technologies

New technologies such as precision farming techniques have the potential to improve N use efficiencies (Gotway *et al.*, 1996; Hergert *et al.*, 1996; Redulla *et al.*, 1996). Variable rate maps and/or management zones can be used to improve the accuracy of N fertilizer applications (Delgado and Bausch, 2005; Delgado *et al.*, 2005; Ferguson *et al.*, 1996; Khosla *et al.*, 2002). New technologies can help improve N management by providing information to nutrient managers about the potential N uptake and N status throughout the growing season. Nutrient managers can use this information to develop N management plans that better synchronize N inputs with crop N sinks from preplanting through harvest. Some of these technologies can provide spatial and temporal information during the growing season and help nutrient managers identify areas that are deficient or overfertilized with nitrogen.

Soil samples can be collected using a Global Position System (GPS), then analyzed in a laboratory to provide information about the spatial variability of residual soil inorganic N, SOM, and mineralization potential. There are other technologies such as remote sensing, that can be used to instantly provide information about the N status of large field areas.

Site-specific management zones (SSMZ) can be used to manage N based on yield history, soil color from aerial photographs, topography, and the producer's past management experiences (Fleming *et al.*, 1999). SSMZ can be used to develop an N management plan that considers the variability in N sinks using realistic yields across the field. Additionally, management zones integrate the potential N from SOM, residual NO_3-N and other sources that are representative of each zone instead of using a yield average.

Recent research has shown that these new technologies can help increase N use efficiencies (Khosla *et al.*, 2002) and reduce NO_3-N leaching (Delgado *et al.*, 2005). The lower yield, sandier, coarser areas, which received greater N applications, had greater leaching losses because the N sink was much lower than areas with higher yields. Applications of N according to management zones or spatial variability of N sinks can increase agronomic N use efficiencies and reduce losses of N to the environment by minimizing NO_3-N leaching (Delgado *et al.*, 2005; Khosla *et al.*, 2002).

We evaluated the data presented by Delgado *et al.* (2005) and Delgado and Bausch (2005). We estimated excessive N applications with the following formula:

$$\mathrm{ENFA} = [\mathrm{NFA} - (\mathrm{CU} - \mathrm{CUWF})] \quad (1)$$

where ENFA is excessive N fertilizer application, NFA is N fertilizer applied, CU is aboveground crop uptake at the given fertilizer rate, and CUWF is aboveground crop N uptake by plant without fertilizer.

This definition of excessive N fertilizer, based on the net N uptake from the added N fertilizer, was correlated with the sum of simulated residual soil

NO_3-N and NO_3-N leaching by zones ($P < 0.01$). The areas of the fields with higher sand content (low-productivity zones) had lower residual soil NO_3-N content (Fig. 13). It is clear that any N applications greater than the crop N uptake will increase NO_3-N leaching and NO_3-N available to leach across all of the zones (Fig. 12).

Figure 12 is in agreement with Andraski *et al.* (2000), who defined excessive N fertilizer application as the applied N fertilizer rate minus the economically optimum N fertilizer rates, correlated with soil water NO_3-N concentrations. Our definition of excessive N application calculates the N that is available for loss to the environment, accounting for a site-specific N uptake of zero N fertilizer, and assessing all sources of N except N inputs from fertilizer or manure. Figure 12 is also in agreement with Pratt (1979), who reported that we cannot completely eliminate NO_3-N leaching losses. Delgado *et al.* (2006, 2008a) N index ranks NO_3-N leaching losses from the system as very low ($\leq$28 kg N ha^{-1} y^{-1}), low ($>$28 $\leq$ 56 kg N ha^{-1} y^{-1}), medium ($>$56 $\leq$ 112 kg N ha^{-1} y^{-1}), high ($>$112 $\leq$ 168 kg N ha^{-1} y^{-1}), and very high ($>$168 kg N ha^{-1} y^{-1}).

Delgado *et al.* (2005) and Delgado and Bausch (2005) showed that using SSMZ and remote sensing in conjunction with reducing excessive N fertilizer applications can significantly reduce NO_3-N leaching losses. Delgado *et al.* (2005) concluded that spatially variable N management based on productivity zones produces less NO_3-N leaching than uniform strategies while maintaining maximum yield. They estimated that we can cut NO_3-N leaching losses by 25% during the first year by using an SSMZ-based nutrient management plan.

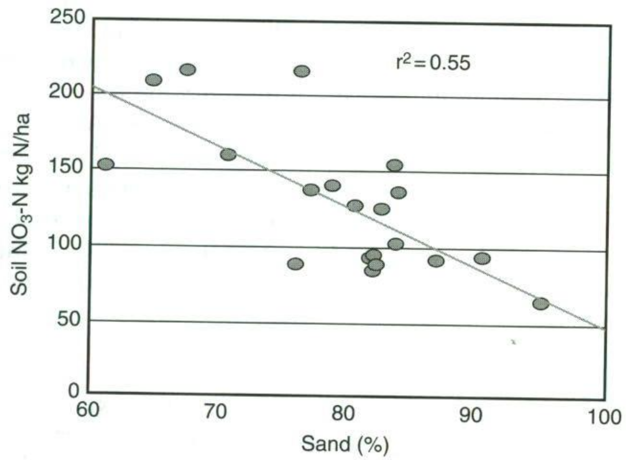

Figure 13 Correlation between the residual soil NO_3-N in the top 1.5 m of soil with the respective sand content at each site during the 2000 growing season (from Delgado and Bausch, 2005).

Remote-sensing techniques can be used to monitor spectral reflectance to determine crop N status, including deficiency levels that may reduce yields (Al-Abbas *et al.*, 1974; Stanhill *et al.*, 1972; Thomas and Gausman, 1977). Remote sensing has allowed the development of reflectance indices used to monitor N status during the growing season, such as the N Reflectance Index (NRI) by Bausch and Duke (1996), and the Normalized Difference Vegetation Index (NDVI) by Tucker (1979) and Wood *et al.* (1999). These techniques and indices can quickly provide *in situ* information to help determine the need for N applications (Raun and Schepers, 2008).

These N indices and remote-sensing techniques have allowed us to determine spatially variable N status across fields (Bausch *et al.*, 1996; Blackmer *et al.*, 1996; Franzen *et al.*, 1999; McMurtrey *et al.*, 1994; Raun and Schepers, 2008; Scharf *et al.*, 2002). For example, crop N information gathered with remote sensing was used to cut N applications to 50% of traditional application rates (Bausch and Delgado, 2003), reducing NO_3-N leaching losses by 47% (Delgado and Bausch, 2005).

Other relatively new tools include chlorophyll meters and portable electrodes that can help monitor N levels during the growing season to further increase the N use efficiency through split N applications (Follett *et al.*, 1992; Schepers *et al.*, 1992a,b; Turner and Jund, 1991). Chlorophyll readings can be compared with N application rates to identify areas requiring additional N applications (Schepers *et al.*, 1992a,b). Delgado *et al.* (2001b) reported a correlation between the leaf chlorophyll readings and potato tuber yield and quality. These tools have the potential to be used to determine N status and the need for N fertilizer applications, especially under irrigated systems.

Another relatively new method is the use of field test strips to assess N status by determining sap NO_3-N concentration for vegetables (Prasad and Spiers, 1984; Scaife and Stevens, 1983; Williams and Maier, 1990) and small grains (Papastylianou, 1989). Portable NO_3-N ion-selective instruments can also be used to measure sap NO_3-N concentrations for vegetables (Errebhi *et al.*, 1998; Hartz *et al.*, 1994; Kubota *et al.*, 1996, 1997; Westcott *et al.*, 1993) and winter cover crops (Delgado and Follett, 1998).

Collecting plant samples for laboratory testing is a more traditional method for determining N status. This approach may require additional time compared to remote-sensing techniques, chlorophyll meters, portable electrodes, and field strips, because of the time needed to run the samples in the laboratory to get a recommendation about N status. Laboratory results can also be combined with SSMZs and precision farming techniques if the samples are collected using Global Position Systems.

An example of laboratory-based tissue analysis is the potato petiole NO_3-N test (King *et al.*, 1999). The presidedress soil NO_3-N test (PSNT) can also be used to monitor crop N status (Bundy and Meisinger, 1994). The PSNT is commonly used in the Northern Corn Belt and the northeastern United States to assess the available soil NO_3-N pool to identify if N levels are sufficient and/or to provide a basis for sidedress fertilizer N recommendations (Bundy and Meisinger, 1994). The PSNT can help increase N use efficiencies and lower NO_3-N leaching potential (Durieux *et al.*, 1995; Guillard *et al.*, 1999).

2.2.5. Models and index

An N budget based on an estimation of the percentage of applied N taken by the crop could be used to conduct a quick assessment of the potential for N losses (Bock and Hergert, 1991). However, with the Bock and Hergert (1991) N use efficiency index, there is no information about what may happen to the N that is not absorbed by the crop. Shaffer and Delgado (2002) discussed advantages and disadvantages of several Nitrogen Indexes that can be used to assess N management. A Nitrogen Index that considers N losses to the environment could potentially be used to conduct an assessment of how N management practices are affecting N losses (Delgado *et al.*, 2006, 2008a). This new qualitative/quantitative N index can be joined to GIS to discern practices that have very low, low, and medium potential for N losses from practices that have high and very high potential risk for these losses (De Paz *et al.*, 2008).

Although N indexes could be used to conduct quick assessments of N losses, an NTT requires a more robust approach such as the use of an N model that can integrate detailed layers of information about soil–crop–hydrologic systems to assess losses of nitrogen from the nitrogen cycle (Delgado *et al.*, 2008c).

There are several national and international models that can be used to assess N losses to the environment. Examples of these models include the Nitrate Leaching and Economic Analysis Package (NLEAP) (Delgado *et al.*, 1998; Shaffer *et al.*, 1991), the Crop Estimation through Resource and Environmental Synthesis (CERES) (Ritchie *et al.*, 1985), Erosion Productivity Impact Calculator (EPIC) (Williams *et al.*, 1983), Nitrogen Tillage Residue Management Model (NTRM) (Shaffer and Larson, 1987), Root Zone Water Quality Model, RZWQM (Shaffer *et al.*, 2000), LEACHM (Wagenet and Hutson, 1989), and the Great Plains Framework for Agricultural Resource Management (GPFARM) (Ascough *et al.*, 2001). For additional information on other national and international models that simulate N dynamics and transport, see Shaffer *et al.* (2001). The initial prototype of the NTT used NLEAP (Delgado *et al.*, 2008c; Gross *et al.*, 2008), but if nitrogen trading markets are more widely implemented throughout the world, and/or nitrogen trading is integrated with the trading of potential carbon sequestration credits, it is possible that we could see a series of other NTTs developed in the near future for national and international users.

2.2.6. Identifying and managing spatial and temporal variability

There are new advances in software that can be used to identify spatial variability (Berry, 2003a,b, 2007a,b). Recent advances in current geospatial research have been refocusing on data structure and analysis (Berry, 2007a). Delgado and Berry (2008) reported on how to identify spatial patterns and to manage spatial variability with precision conservation to reduce environmental impacts.

Watershed models such as the Agricultural Non-Point Source Pollution (AGNPS) model (Young *et al.*, 1987) and the Soil and Water Assessment Tool (SWAT) model (Arnold *et al.*, 1993) can be used to assess erosion losses. These models are also being used to assess nutrient losses. The assessment of chemical movement, runoff, and erosion was also conducted using the Agricultural Management Systems (CREAMS) (Smith and Williams, 1980).

Renschler and Lee (2005) used three models and GIS to evaluate the effects of best management practices. The models used were the Water Erosion Prediction Project (WEPP), the Geospatial interface for WEPP (GeoWEPP), and SWAT. Bonilla *et al.* (2007) used the Precision Agricultural-Landscape Modeling System (PALMS) and reported that PALMS can evaluate the effects of local soil properties and microtopography on changes in soil detachment and deposition across short distances and has the capability to quantify a series of spatial and temporal parameters. Modeling can be used to assess spatial erosion and N losses across the environment. There is also potential to use the NLEAP GIS 4.2 prototype to assess spatial N losses at the field level and/or nitrogen trading at the field level (Delgado *et al.*, 2008b,c). For additional details about precision conservation and identifying and managing spatial and temporal variability, see Delgado and Berry (2008).

2.2.7. Rotation of crops

Nitrogen management can be improved with crop rotations and more efficient crop varieties. Deeper-rooted crops can be rotated into shallower-rooted systems to increase the N use efficiency of the system. The deeper-rooted crops recover NO_3-N from groundwater, minimizing the net NO_3-N leaching from the system and contributing to water conservation (Delgado, 1998, 2001). Deeper root depth was correlated with less NO_3-N leaching, greater NO_3-N mining, and higher N use efficiencies (Delgado, 1998, 2001; Delgado *et al.*, 2006).

Some researchers have reported on the potential of winter cover crops to reduce NO_3-N leaching (Delgado, 1998; Meisinger *et al.*, 1991; Shipley *et al.*, 1992). The inclusion of winter cover crops in rotations can increase system N use efficiencies, not only by recovering N from the previous crop but also by reducing N losses from the next crop (Delgado, 1998, 2001).

Scavenger cover crops can increase N cycling by increasing the N sink during the fallow period (Delgado, 1998; Meisinger *et al.*, 1991; Shipley *et al.*, 1992). Delgado *et al.* (2007) reported that summer cover crops with limited irrigation can increase the N sink during the fallow season, significantly increase N use efficiency, and improve yield and quality of the following crop. Multiple crops per year can also involve grasses, which may be harvested multiple times and which may help increase the potential for N trading.

Adding a legume to the crop rotation can further increase the N use efficiency of the systems and reduce N losses (Meisinger and Delgado, 2002; Randall *et al.*, 2008). Because leguminous crops can fix N from the atmosphere they require lower or zero N inputs, which, combined with the residue N cycling to the following crop, reduces NO_3-N leaching potential even more (Kanwar *et al.*, 1997; Randall *et al.*, 1997). These studies show the potential for NO_3-N leaching reduction in tile systems when the leguminous crops are included in grain rotations. In Virginia, this practice increased the potential for N trading over the baseline, with the potential to trade N (Delgado *et al.*, 2008c). The savings in N_2O were up to 4 kg N ha^{-1}, which generated the potential to trade 500 kg C ha^{-1} as carbon sequestration equivalents (Delgado *et al.*, 2008c; Lal *et al.*, 2009).

It is clear that quantification of N losses to the environment is difficult; however, we can use isotopic ^{15}N techniques to assess N losses. Delgado *et al.* (2004) developed a crop residue exchange method to assess N cycling, fate and losses from crop residues on a large plot scale that was used for cover crop residue exchange studies in Colorado and the Pacific Northwest (Collins *et al.*, 2007). The results from these studies show that the 72 and 58% N recovered (soil and plant) from fertilizer in Colorado and the Pacific Northwest, respectively, were much lower than the 85 and 95% recovered from crop residue. These cover crop studies are important in that they point out that N losses from fertilizer are two times greater than N losses from crop residues (Delgado *et al.*, 2004, 2007). In other words, cover crops not only increase the system N use efficiency and NO_3-N mining from underground water, but also increase N cycling to the subsequent crops, leading to higher N use efficiencies and fewer N losses to the environment than fertilizer inputs. Delgado *et al.* (2010) reported that these unique ^{15}N crop residue exchange studies and simulations conducted with the DAYCENT model showed that the N losses from crop residue are much lower than those from inorganic N fertilizer, including lower emissions of N_2O and NO_3-N leaching. Delgado *et al.* (2010) recommended the use of lower coefficients for N_2O emissions from crop residue, especially if they have a high C/N ratio (>30).

Delgado *et al.* (2007) reported that summer cover crops with limited irrigation are being used by farmers in Colorado. If farmers were to implement a summer cover crop with limited irrigation program more widely than is done currently, the potential savings in reactive N to the environment across this region could be as much as 300,000 kg N y^{-1} for approximately every 94 center irrigated pivots (about 60 kg N ha^{-1} y^{-1}). The savings in N that would then be available to trade would also generate 860,000 kg C sequestration equivalents due to direct and indirect reductions in emissions of N_2O (Figs. 10 and 11, Table 1). Results from these studies are in agreement with the Al-Sheikh *et al.* (2005) report that increasing rotation of deep-rooted crops and incorporation of crop residue increase the N sequestration in this region. In addition to the advantages just described, farmers would also benefit from tremendous savings in irrigation water.

2.2.8. Summary of N inputs

Crop rotations, lower N inputs, split N applications, leguminous crops, cover crops, and modified methods of applications (such as incorporation of manures), and other practices all can be used to reduce N losses the environment and increase N savings (Delgado *et al.*, 2008c). These N savings may be even more substantial depending on the practice(s) used and soil combinations present (Delgado *et al.*, 2008c). An Internet-based or stand-alone NTT can be used to assess potential N savings and the potential to trade N in conservation markets (Delgado *et al.*, 2008c; Gross *et al.*, 2008).

2.3. Transformations and pathways for reactive and total nitrogen losses

Several scientists have reported that it may be possible to use denitrification as a method to reduce the losses of reactive N to the environment (Hey, 2002; Hey *et al.*, 2005; Hunter, 2001; Mosier *et al.*, 2002). This can be achieved by adding a carbon source to the system (Mosier *et al.*, 2002), strategically placing denitrification traps (Hunter, 2001), strategically managing water levels of drainage systems (Strock *et al.*, 2007), and strategically locating wetlands to increase denitrification and removal of NO_3-N from surface water (Hey, 2002; Hey *et al.*, 2005). This strategic use of denitrification-based management practices is another example of how precision conservation that considers spatial and temporal variability can be used to reduce N transport in the environment and increase N trading potential.

Since some scientists recommend denitrification as a positive pathway for removing NO_3-N from surface and groundwater flows, we defined the NTT as the quantification of the mathematical difference between a base scenario and a new N management scenario by adding individual pathways of the N cycle. Since denitrification (N_2-N) loss has been reportedly beneficial in some cases by reducing the effects of reactive N on the environment (Hey, 2002; Hey *et al.*, 2005; Hunter, 2001; Mosier *et al.*, 2002), we calculated the NTT-DNL_{reac} using Eqs. (2)–(7).

The following equations are used to calculate reactive N losses, which include nitrate leaching (ΔNO_3-N, Eq. (2)), nitrous oxide losses (ΔN_2O-N, Eq. (3)), ammonia volatilization (ΔNH_3-N, Eq. (4)), surface N transport not connected to soil erosion (ΔN_{st}, Eq. (5)), surface N transport caused by soil erosion (ΔN_{er}, Eq. (6)), and NTT-DNL_{reac} (Eq. (7)):

$$\Delta NO_3 - N = NO_3{-}N_{bms} - NO_3{-}N_{nms} \tag{2}$$

$$\Delta N_2O - N = N_2O - N_{bms} - N_2O - N_{nms} \tag{3}$$

$$\Delta NH_3 - N = NH_3{-}N_{bms} - NH_3{-}N_{nms} \tag{4}$$

$$\Delta N_{st} - N = N_{st}{-}N_{bms} - N_{st}{-}N_{nms} \tag{5}$$

$$\Delta N_{er} = N_{er}{-}N_{bms} - N_{er}{-}N_{nms} \tag{6}$$

$$\begin{aligned} NTT - DNL_{reac} = {} & \Delta NO_3 - N + \Delta N_2O - N + \Delta NH_3 \\ & -N + \Delta N_{st} + \Delta N_{er} \end{aligned} \tag{7}$$

If the nutrient managers are also interested in N use efficiencies in the cropping system, they will want to know the effects of nonreactive N losses due to denitrification. To calculate total N losses, Eq. (8) is used to calculate N_2-N denitrification (ΔN_2-N) and Eq. (9) is used to calculate the NTT difference in total N losses (NTT-DNL_{tot}). For Eqs. (2)–(9), *bms* refers to the base management scenario, and *nms* refers to the new management scenario:

$$\Delta N_2 - N = N_2{-}N_{bms} - N_2{-}N_{nms} \tag{8}$$

$$NTT - DNL_{tot} = NTT - DNL_{reac} + \Delta N_2 - N \tag{9}$$

Some users will be interested in trading N in air quality markets as carbon sequestration equivalents (Delgado *et al.*, 2008c; Lal *et al.*, 2009). The carbon sequestration unit equivalents earned through the reduction of N_2O-N losses to the atmosphere can be estimated with Eq. (10) (ΔN_2O-N$\times$132.8). The International Panel on Climate Change (IPCC) methodology also accounts for indirect N_2O emissions from reactive N losses to the environment. The IPCC's methodology assumes that 30% of fertilizer N input is leached and/or lost as runoff and that 0.75% of these losses are emitted as N_2O-N (Eggleston *et al.*, 2006; Houghton *et al.*, 1992). Additionally, the IPCC methodology assumes that 10% of the N fertilizer (20% of the manure N) is lost through NH_3-N/NO_x-N volatilization and that 1.0% of these losses are also emitted indirectly as N_2O-N (Eggleston *et al.*, 2006). The indirect savings in carbon sequestration equivalents due to the reduction in direct N_2O losses are estimated with Eq. (11). The total savings in carbon sequestration equivalents due to the reduction in direct and indirect N_2O losses are estimated with Eq. (12):

$$\Delta DCO_2{-}C_{seN_2O} = \Delta N_2O - N \times 310 \times 0.2727 \times 1.571 \tag{10}$$

$$\begin{aligned}\Delta ICO_2{-}C_{seN_2O} = &[((\Delta NO_3 - N + \Delta N_{st} - N + \Delta N_{er}) \\ &\times 0.0075 \times 310 \times 1.571) \\ &+(\Delta NH_3 - N \times 0.01 \times 310 \times 1.571)] \times 0.2727\end{aligned} \tag{11}$$

$$\Delta TCO_2{-}C_{seN_2O} = \Delta DCO_2{-}C_{seN_2O} + \Delta ICO_2{-}C_{seN_2O} \tag{12}$$

2.3.1. Gaseous pathways

There are several gaseous pathways by which N gases may be emitted from soils (Fig. 1). Researchers have conducted *in situ* field and laboratory studies to measure the effects of management practices on emissions of N gases and how management of gaseous losses affects N use efficiencies. One of the most important pathways for N loss is denitrification (Figs. 1 and 3). The acetylene technique is based on the discovery by Federova *et al.* (1973) that the reduction from N_2O to N_2 in the denitrification process can be inhibited with acetylene. Isotopic ^{15}N labeled N has been used to trace the effects of management on denitrification.

The process of denitrification has been studied very closely by Firestone and Davidson (1989), Hutchinson (1995), and Mosier and Klemedtsson (1994), among others. Biogeochemical reactions of nitrification and denitrification drive emissions of $N_2O/NO/N_2$ (Fig. 3). Although emissions of N_2O are minimal and reported to be an average 1% of the applied N fertilizer (Eggleston *et al.*, 2006), the losses of N_2 due to denitrification could be significant (Meisinger and Randall, 1991; Peoples *et al.*, 1995).

Denitrification potential has been correlated with surface texture and drainage characteristics by several scientists. Peoples *et al.* (1995) reported that potential denitrification for poorly drained clay soils was 35%, seven times higher than the 5.5% for the well-drained sandy soils. Similarly, Meisinger and Randall (1991) reported that potential denitrification was 25–55% for poorly drained soils with over 5% SOM, while the potential denitrification was about 6–20% for the well-drained soils.

Mosier *et al.* (2002) reported that we can manage denitrification with water and nitrogen management practices and carbon inputs. Nitrification inhibitors (Bronson and Mosier, 1993; Freney *et al.*, 1992) and controlled-release fertilizers (Delgado and Mosier, 1996; Shoji and Gandeza, 1992; Shoji and Kanno, 1994; Shoji *et al.*, 2001) can be used to further reduce N_2O emissions. Mosier *et al.* (2002) recommended that the best practice for reducing N_2O emissions is to develop a management plan that increases N use efficiencies and reduces N inputs.

The Global Warming Potential over a 100-year time frame for N_2O is about 310 (USEPA, 2007; http://www.epa.gov/OMS/climate/420f05002.htm#global). In other words, a management practice that reduces N_2O emissions by 1.0 kg N_2O-N is equivalent to the sequestration equivalents of 132.8 kg CO_2-C. An NTT can be used to evaluate the effectiveness of these techniques for reducing N_2O emissions and the resultant ability to trade these reductions as carbon sequestration equivalents in air quality markets (Delgado *et al.*, 2008c). Finer soils with greater denitrification potential and greater N_2O emission potential offer an advantage for trading carbon sequestration equivalents, particularly under irrigated systems, because greater reductions of N_2O emissions can be achieved than with coarser soils (Delgado *et al.*, 2008c). NTT results show that practices that match the N application with N uptake or reduce excessive N applications mitigate denitrification, N_2O and NO_3-N leaching losses and increase the potential for N trading and trading of carbon sequestration equivalents (Eqs. (10)–(12)).

Mosier *et al.* (2002) reported that management practices that increase N use efficiencies, such as using N budgets to avoid overapplication, using the right N source with respect to water management, splitting N into multiple applications, improving water management, using source types to reduce denitrification, and other N management methods can lead to reduced denitrification losses. Management of soil denitrification will also be correlated with management of soil oxygen concentrations and water-filled pore space (e.g., soil water content) (Freney *et al.*, 1992; Gilliam and Boswell, 1984; Hey *et al.*, 2005; Linn and Doran, 1984; Meisinger and Randall, 1991; Mosier *et al.*, 2002; Peoples *et al.*, 1995; Steenvoorden, 1985). Additionally, management of soil denitrification will also be correlated with management of carbon inputs (Firestone and Davidson, 1989; Hunter, 2001; Meisinger and Randall, 1991; Mosier *et al.*, 2002; Peoples *et al.*, 1995; Weier *et al.*, 1993, 1994).

Management of denitrification can be used as a mitigation alternative to reduce the off-site transport of N across the environment. We could use management of oxygen levels in soils by managing water levels to increase denitrification rates for the removal of NO_3-N, thereby reducing its transport in the environment (Gilliam and Boswell, 1984; Hey *et al.*, 2005; Hunter, 2001; Mosier *et al.*, 2002; Steenvoorden, 1985). Alternatively, we could add carbon sources to increase dentrification rates of nitrate that has been leached out of the system (Hunter, 2001).

Delgado *et al.* (2008c) recommended that denitrification should not be accounted for when evaluating the potential reduction of reactive N losses to the environment, and that any methods that reduce the NO_3-N transport at a farm or field level should be counted as a practice that reduces the transport of reactive N losses to the environment. Thus, if a management practice increases denitrification losses and reduces the transport of reactive NO_3-N, the new practice will be basically credited with savings as far as reducing potential N losses of reactive N over the baseline scenario. However, a full analysis should consider N_2O emissions, since this management practice may also increase N_2O emissions under a higher denitrification potential, depending on the oxygen levels (Mosier *et al.*, 2002). An NTT could provide the advantage of conducting a mass balance analysis of both pathways simultaneously to determine if the reduction of NO_3-N transport due to denitrification may increase N_2O emissions.

At a watershed level, the concept of nutrient farming proposed by Hey *et al.* (2005) is a very valuable one and can serve as a key precision conservation practice (Delgado and Berry, 2008). However, it remains to be sorted out how nitrogen trading systems will credit farmers for reducing the transport of NO_3-N out of their fields at an upstream watershed while simultaneously crediting a nutrient harvesting farm downstream without double accounting. We suggest that farmers who reduce the NO_3-N transport may get a credit at a farm level, while the implementation of a wetland area or riparian forest downstream may be credited with the balance between NO_3-N transport into the system and NO_3-N coming out, since these systems will serve as potential filters for NO_3-N. However, the effect of the denitrification on potential N_2O emissions, the emissions of other gases such as methane, and even on carbon sequestration may also have to be sorted out. Other critical factors such as distance to water bodies (like streams and rivers) would need to be considered as well, but will not be covered in this chapter. A full analysis for the nutrient harvesting wetland may be needed to determine the balance between carbon and nitrogen pools.

Another important form in which N is lost to the atmosphere is NO_x, a pathway that does not result in as many losses as N_2, but generally presents N losses much greater than N_2O. For example, it has been reported that the 1.3 kg NO_x ha^{-1} y^{-1} lost from a Colorado short grass steppe was about 10 times greater than the N_2O emissions (Martin *et al.*, 1998) and was driven mainly by N mineralization. These results were in agreement with Hutchinson (1995), who reported that NO is formed in the denitrification process, but is not considered a major product of denitrification because of the combined effect of high water content restricting NO diffusion into the atmosphere and the further reduction of NO into N_2O and N_2.

Another significant pathway for gaseous losses of N is NH_3-N volatilization from fertilizers and animal wastes that contain urea and NH_4-N (Peoples *et al.*, 1995). Peoples *et al.* (1995) reported that losses due to NH_3-N volatilization can be significant in every part of the world, especially in sensitive systems such as flooded rice in Australia, China, India, and the Philippines (45–78%) and sugarcane fields in Australia (47–61%). However, these losses can be significantly reduced through proper management. Studies have indicated that higher levels of NH_3-N volatilization correlate with higher pH. For example, the volatilization of urea in flooded rice was reported to be about 9%, much lower than the 30% observed when the site was in a flooded calcareous soil (Peoples *et al.*, 1995). Peoples reported that small grain systems such as barley, sorghum, and wheat usually receive broadcast applications and incorporation, with reported decreases in losses via NH_3-N volatilization (<20%).

Meisinger and Randall (1991) also found that lower pH and incorporation of N reduce NH_3-N volatilization losses. They reported that the NH_3-N volatilization of unincorporated urea for soils with a pH of 7.0 and above could be as high as 20%, compared to the 5% reported when the urea was broadcast in a humid climate. Similarly, volatilization of NH_3-N from manures was much lower when the manures were incorporated. Management practices that incorporate N sources will help reduce NH_3-N emissions and will result in greater potential to trade these N savings. For more research related to NH_3-N emissions from agricultural systems, see Fox *et al.* (1996), Freney *et al.* (1981), Sharpe and Harper (1995), and Wood *et al.* (2000).

2.3.2. NO_3-N leaching pathways

The background NO_3-N concentrations of natural systems have been reported to be lower than 2 mg NO_3-N l^{-1} (Hallberg, 1989). It has been reported in studies throughout the world that increases of N inputs and changes in land use patterns have been correlated with increases of background NO_3-N concentrations for underground water (De Paz *et al.*, 2008; Fletcher, 1991; Hallberg, 1989; Juergens-Gschwind, 1989; Wylie *et al.*, 1994). These increases in NO_3-N concentrations are increasing concerns about the environment across national and international communities, particularly because of the potential for cases of methemohemoglobia (or "blue baby syndrome") that can occur from drinking water with NO_3-N concentrations greater than the 10 mg NO_3-N l^{-1} (Follett and Walker, 1989; Follett *et al.*, 1991). Additionally, the losses of N resulting from NO_3-N leaching have been established as an important indirect source of emissions of N_2O to the atmosphere (Eggleston *et al.*, 2006; Houghton *et al.*, 1992).

Fortunately, it may be possible to increase N use efficiencies and reduce NO_3-N leaching losses to the environment and even mine NO_3-N from ground waters (Delgado, 1998; Delgado *et al.*, 2001a). Nitrate leaching is a function of water leaching and the concentration of NO_3-N at the time of the water movement outside the root zone. There cannot be NO_3-N leaching without water leaching, so water management is essential to reducing these N losses. Meisinger and Delgado (2002) described the principles that can be used to reduce NO_3-N leaching; see their discussion for additional information on this subject.

Pratt (1979) reported that it is almost impossible to eliminate NO_3-N leaching because of weather and irrigation water inputs. However, management decisions are the primary factors that help minimize NO_3-N leaching losses, even in irrigated sandier coarse textured sites, which are the most susceptible to nitrate leaching (Delgado, 2001; Shaffer and Delgado, 2002; Fig. 13). Best management practices that significantly reduce NO_3-N leaching losses also increase the potential for N trading, with N loss savings up to 100 kg N ha^{-1} (Delgado *et al.*, 2008c).

2.3.3. Erosion N loss pathways

Erosion's negative effects are not limited to off-site transport of N to water bodies. Erosion can also reduce yields, thereby reducing the N sink that is generated with higher yields, and increasing the potential for N losses related to reduced agricultural production. In drier regions, the main mechanism for erosion-based surface transport of N is wind. Wind can detach fine particles and carry them to other areas where they are deposited via dry or wet deposition (Skidmore *et al.*, 1970). Researchers who have studied wind erosion have reported that this transport can affect off-site surface water and groundwater sources because the transported particles can carry SOM and inorganic NO_3-N and NH_4-N, all of which can be sources of NO_3-N, which can reach groundwater through leaching (Cihacewk *et al.*, 1993). In humid and irrigated systems, erosion forces are primarily due to water runoff. Rain can affect bare soil surfaces and can break up soil aggregates, facilitating the transport of dissolved chemicals and/or loose particles that can impact water bodies (Foster *et al.*, 1982; Truman *et al.*, 2001). Compacted areas, and areas with low permeability will have greater NO_3-N runoff and greater total N discharge (Rochester *et al.*, 1994).

The hydrologic cycle, soil type, soil cover, and slope are factors that can affect the rate of soil erosion. Initial efforts in assessing spatial erosion impacts by accounting for topography and other parameters were reported by Wheeler (1990), Mitasova *et al.* (1995), Desmet and Govers (1996), Siegel (1996), Mitas *et al.* (1997), and Wang *et al.* (2000). Other important initial steps were taken by Wischmeier and Smith (1965) using the Universal Soil Loss Equation (USLE) to calculate average soil losses on slope sections; their work was expanded to a watershed scale by Foster and Wischmeier (1974), Williams and Berndt (1972), and Wilson (1986). Currently we have more advanced models that account for spatial erosion variability using GIS and Digital Elevation Models (DEMs) (Desmet and Govers, 1996). Some of the models used to evaluate watersheds are the AGNPS model (Young *et al.*, 1987) and the SWAT model (Arnold *et al.*, 1993).

Keeping the soil covered with residue management or minimum tillage is an essential concept that can be used to help reduce off-site N transport (Holt, 1979). Cover crops can help maintain the soil cover and significantly reduce wind and water erosion (Bilbro, 1991; Dabney *et al.*, 2001; Delgado *et al.*, 1999). Efforts to identify advantages of management practices and/or erosion hot spots in a watershed have been conducted recently by Secchi *et al.* (2007), Renschler and Lee (2005), Qiu *et al.* (2007), Dosskey *et al.*

(2005, 2007), and Bonilla *et al.* (2007), among others. However, there is still the need to develop more robust models that will include all the biogeochemical pathways for nitrogen, phosphorous, and carbon with surface and underground routing across a watershed to connect fields with drainage channels, wetlands, and riparian zones in three dimensions (Delgado and Berry, 2008).

For the purposes of this study, we are limiting the focus of our discussion to the potential to use an NTT at a field level; the interaction of nitrogen and carbon and/or other trace gases such as methane will not be covered here. Additionally, the potential for lag impact of N losses to water bodies depending on the hydrology and pathways that may affect the transport of N across the soil profile or drainage systems will also not be examined. Instead, our discussion is intended to evaluate the effects of management practices at the field level. We propose that if the N losses are reduced from all nonpoint sources at a field level, by using principles that increase N use efficiencies that maintain viable agricultural production while reducing N inputs, that the off-site transport of N in the environment will also be significantly reduced.

2.4. Nitrogen management and long-term effects on nitrogen pools

Delgado and Follett (2002) recommended that carbon management should be an integral part of nutrient management because of its positive effects on porosity, available water-holding capacity, cation exchange capacity, and the reduction of toxicity from certain elements. They reported that as management practices increase SOM, the required N inputs are reduced because of the higher N use efficiencies generated by increased N cycling, reducing the potential for NO_3-N leaching and N losses (Fig. 6). Additionally, SOM is important because of its contributions to positive soil physical and chemical characteristics that improve soil productivity and nutrient use efficiency.

As reported by Delgado and Follett (2002), the management of N and C is crucial because it can increase SOM, consequently increasing N use efficiencies and reducing N losses. For example, Vigil *et al.* (2002) reported that average N mineralization was about 45 kg N ha^{-1} for every 1% of SOM. If we increase the SOM content from 1% to 3%, we can increase N release from SOM from 45 to 135 kg N ha^{-1}. This 1–3% increase in SOM will increase the amount of N available for crop uptake and will reduce the need for N inputs.

If we increase the addition of carbon to soils with manures and crop residues we can increase or maintain the amount of SOM, even in cultivated systems (Al-Sheikh *et al.*, 2005; Campbell and Zentner, 1993; Havlin *et al.*, 1990; Larson *et al.*, 1972; Rasmussen *et al.*, 1980). Havlin *et al.*

(1990) reported that increases in the amount of crop residue returned to the soil can increase SOM-C and SOM-N. They reported that SOM-C and SOM-N increases were greater under no till but still showed increases correlated with the amount of crop residue returned to the soil under conventional tillage. Other scientists have shown that increased applications of N fertilizer that result in higher yields, also produce more crop residue, which increases SOM-C and SOM-N (Campbell and Zentner, 1993; Havlin *et al.*, 1990; Rasmussen *et al.*, 1980).

Cropping systems that reduce soil erosion also reduce the losses of SOM-C, SOM-N, and other nutrients from the system (Al-Sheikh *et al.*, 2005; Black and Tanaka, 1997; Hussain *et al.*, 1999; Lal, 2000). Increasing cropping system intensity with fewer fallow periods helps return greater amounts of crop residue and increase SOM-C and SOM-N levels (Black and Tanaka, 1997; Peterson and Westfall, 1997; Rasmussen and Rohde, 1988).

2.5. Relationships: Carbon and nitrogen sequestration and emissions of N_2O

Cropping system carbon sequestration is correlated with nitrogen sequestration (Al-Sheikh *et al.*, 2005; Black and Tanaka, 1997; Hussain *et al.*, 1999; Lal, 2000). Because N management across cropping systems is connected to the global cycle, management that reduces emissions of N_2O helps reduce global warming potential. One benefit of carbon sequestration is that it also contributes to N sequestration, greater N cycling and helps reduce the need for N inputs, which reduces N losses to the environment (Fig. 6). Additionally, a 1 kg N_2O-N reduction is equivalent to the sequestration equivalent of about 132.8 kg CO_2-C in terms of potential global warming effects.

Nitrogen can be directly sequestered in soils (Al-Sheikh *et al.*, 2005; Havlin *et al.*, 1990); however, this does not necessarily mean that N losses to the environment have been reduced simply by sequestering N in the soils. In fact, excessive manure N applications can sequester N in soils, yet still contribute to excessive N losses to the environment (Delgado *et al.*, 2008c). Another way to sequester N that still results in significant N losses to the environment is to use a no-till system with excessive N inputs. Management is the key to maximizing carbon and N sequestration in manure and no-till systems while minimizing N losses to the environment.

Doran *et al.* (1999) reported that all strategies used to improve soil quality were also correlated with soil organic carbon. Several scientists have examined the correlation of soil carbon with improvement of the soil's physical, chemical, and biological properties, including soil porosity, available water-holding capacity, cation exchange capacity, nutrient cycling, toxicity reduction, and contributions to higher yields and economic returns (Delgado and Follett, 2002; Doran and Jones, 1996; Lal, 1995, 1997, 1999; Stevenson, 1982).

We acknowledge that to conduct a true energy balance assessment one must consider machinery's fuel use and CO_2 emissions, as well as other factors that make up the complexity of a cropping system; however, that is not the goal of this chapter. It is important to keep in mind, though, that most agricultural systems in the world are deficient in N and that this element is required to maximize yields and to help ensure the sustainability of agricultural systems. Additionally, we must continue to improve our best practices to reduce N losses to the environment and to increase resource use efficiencies. Based on these principles, we propose that N management practices can be implemented to help maximize yields, reduce N losses, increase N sequestration in soils, reduce N_2O emission, and generate N savings tradable in air and water quality markets and/or carbon sequestration equivalent markets.

3. New Technologies

3.1. Tier one spreadsheet approaches

Shaffer and Delgado (2002) describe a tier approach to assess N losses to the environment as a simple, quick approach based on qualitative and quantitative rankings. For example, a simple approach with a tool such as the Nitrogen Index (Delgado *et al.*, 2006, 2008a) was described as a Tier One level approach. A more complex model such as NLEAP, based on a daily inputs and larger data sets, was described as a Tier Two approach. The use of a more complex research model, together with data collecting and/or supporting research analysis, was described as a Tier Three level. We suggest that currently ongoing efforts in assessing nitrogen trading using the Tier level method are comparable to previous efforts that used this approach to assess N losses to the environment.

The Nitrogen Credit Calculator developed by the World Resources Institute (WRI), Washington, DC, and the NutientNet, a web-based system, can be seen as Tier One efforts. This Tier One level is a web-based or spreadsheet approach to assess the potential for N credits to be traded. These systems developed by the World Resource Institute allow the users to locate their farms within a given watershed, the Kalamazoo River Watershed of Michigan and the Chesapeake Bay Watershed. This is a user-friendly system where the user will answer a series of questions and obtain an estimate of the nutrient reduction credits for adopting a given alternative management strategy. The user could then try to trade these savings by posting them as available for sale on the web site. The WRI in consultation with the Agricultural Workgroup developed a standardized credit estimation spreadsheet program in Excel ©.[1]

3.2. New prototypes: Web-based and stand-alone modeling approaches

The USDA-NRCS and USDA-ARS-SPNR developed a web-based NTT prototype to assess the effects of management practices on N losses to the environment and potential for nitrogen trading (Delgado *et al.*, 2008c) (Fig. 4). A stand-alone prototype that has GIS capabilities was also developed.

The USDA-NRCS–ARS-NTT web-based prototype has an easy-to-use interface where the user selects nitrogen management practices for a given state and the NTT quickly calculates the potential for nitrogen trading when compared to a given baseline. The stand-alone prototype that we are presenting in this chapter also calculates the potential for savings in direct and indirect carbon sequestration equivalents due to best management practices that reduce N losses (Fig. 14).

3.2.1. Soil and climate databases

The NTT is powerful, versatile, and can run with USA soil databases from NRCS (SSURGO) and the NRCS climate databases.

3.2.2. Nitrogen management databases

The NTT has the capability to conduct a large number of simulations simultaneously, allowing multiple users to access the prototype web site. For the stand-alone version, a given user can conduct up to six concurrent evaluations with a given baseline. Additionally, with the stand-alone GIS version the user could compare new management practices to the given baseline for the site-specific field across larger areas of a region. For example, Figs. 11 and 12 show an NTT evaluation across 94 center irrigated pivots that took only a few minutes to run. The NTT currently has a unique database of management scenarios that were developed at the ARS-SPNR unit. These nitrogen management scenarios can be easily expanded to include nitrogen management practices across the USA.

[1] Manufactures and trade names are necessary to report factually on available data, however the USDA or CSU neither guarantees nor warrants the standard of the product; and the use of a given name by the USDA does not imply approval of that product to the exclusion of others that may be suitable.

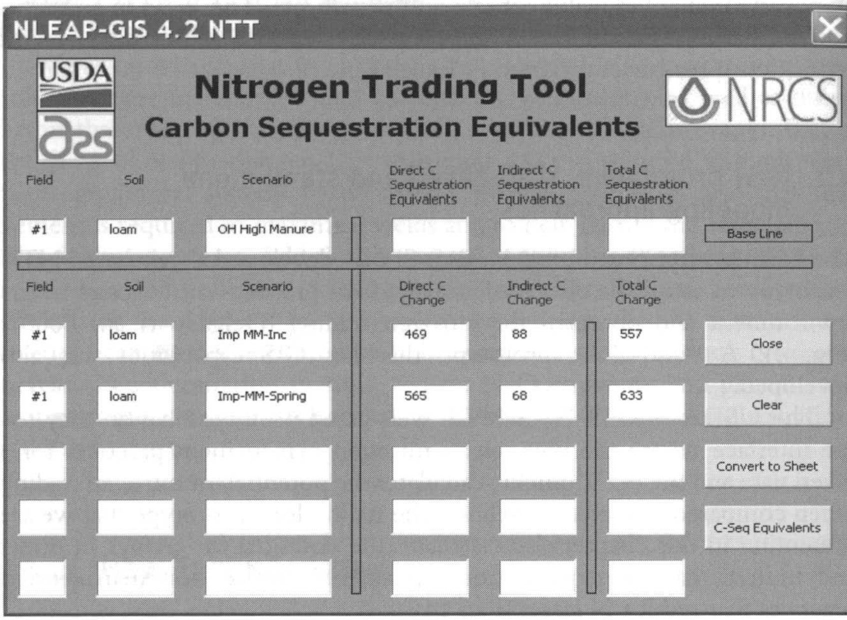

Figure 14 When released, the NTT stand-alone will be able to compare management practices simultaneously to a given baseline in direct and indirect carbon sequestration equivalents.

3.2.3. Long-term evaluations and spatial analysis

The NTT was designed to conduct long-term evaluations (24 years). This feature enables the NTT to have a longer, more robust evaluation. The stand-alone prototype version of the NTT is connected to GIS and runs in Microsoft Version 2003 Excel program© (current plans are to upgrade it to run with the 2010 Version). The prototype can easily import or export NRCS SSURGO soil data in GIS format. Soil data suitable for use with NLEAP-GIS can be downloaded from the NRCS Soil Data-Mart site (http://soildatamart.nrcs.usda.gov/). The NTT is also set to use the NRCS weather data. The user will have to set up the management files for their chosen farm or region using GIS software. Once the GIS files are set, the user could then conduct analysis across the region.

4. Case Scenarios: GIS Trading Tool Concept Evaluations

The NTT uses NLEAP (Delgado *et al.*, 1998; Shaffer *et al.*, 1991) as the simulation model behind the trading tool to conduct simulations using a daily or event-based time interval. Thus, NTT will have the same capabilities and limitations that were described by Shaffer and Delgado (2001) and Delgado and Shaffer (2008) concerning NLEAP. The NTT-GIS prototype conducts an evaluation across a region using point simulations by soil type polygons. Users of this NTT approach need to understand the limitations of the NTT and that there is no mass transfer from polygon to polygon in this Tier Two analysis (see Delgado and Shaffer, 2008; Shaffer and Delgado, 2001; Delgado *et al.*, 2008c). The NTT-GIS tool prototype accounts for surface N transport not connected to soil erosion (Eq. 5) but does not account for N losses due to surface soil erosion (Eq. 6) (see Delgado *et al.*, 2008c). In the majority of our selected scenarios to test this NTT-GIS concept for conservation, the erosion potential was very low (e.g., no-till; slopes lower than 2%) (see Delgado *et al.* 2008b,c). The NLEAP model has been very accurate at the field level and the GIS capabilities and evaluations conducted with NLEAP across a region or field have been validated to adequately assess the N dynamics and losses (Delgado and Bausch, 2005; Hall *et al.*, 2001; Wylie *et al.*, 1994).

4.1. Irrigated systems from dry western US

Delgado *et al.* (2008b) used the NTT-GIS to assess the effects of management across south central Colorado. This GIS evaluation showed that the implementation of best management practices can reduce N losses. Using a potato–potato rotation as a baseline with high N inputs, the use of cover crops with limited irrigation could reduce the N losses and potential to trade N by 53 kg N ha^{-1} (Table 1).

This GIS evaluation showed that the implementation of cover crops with limited irrigation can reduce N_2O emissions and NO_3-N leaching losses increasing the potential for trading direct and indirect C sequestration equivalents (Eqs. (11)–(12), Table 1). There is a potential for trading 189 kg C ha^{-1} in direct and indirect C sequestration equivalents (Table 1). If the user would like to assess specific field comparisons against different baselines for site-specific fields, Figs. 11 and 12 show the potential N savings and C sequestration equivalents potential across the region for a given field.

4.2. No-till systems from north atlantic region

The NTT shows that it is possible to improve management practices by adding a leguminous crop to the rotation and/or winter cover crop with a leguminous crop to a rotation. The inclusion of leguminous crops in a rotation contributed to reduce the nitrogen losses to the environment by about 7 kg N ha^{-1}. The direct and indirect carbon sequestration potential would be 325 kg C ha^{-1} (Table 1).

4.3. Manure operations from midwest region

Better manure management applications and/or applications of manure based on N budgets that did not overapply N, significantly reduced the N losses due to leaching and/or denitrification, and even reduced the amount of N_2O emissions. The NTT results shows that with better manure management there are significant savings in N, leaving a large amount of N available to trade, up to 69 kg N ha^{-1} The direct and indirect carbon sequestration potential to trade was about 595 kg C ha^{-1} (Table 1).

5. Current Applications and Trends

Delgado *et al.* (2008c), Gross *et al.* (2008), and Lal *et al.* (2009) reported that there is potential to integrate nitrogen management with water and air quality markets. This is in agreement with Delgado and Follett (2002) who reported that C management and nutrient cycling should be an integral part of nutrient management plans for maintaining the sustainability of our biosphere. Lal *et al.* (2009) reviewed in detail current trends about air and water-quality trading markets' approaches for improving soil and water conservation, including current trends and tools in nitrogen trading. They discussed opportunities of cap and trade and of voluntary systems for nitrogen trading, and how the interaction between sellers, aggregators, and markets works on current markets and trends for future markets.

5.1. Water quality markets

The nitrogen trading potential for water quality markets looks promising. There are several states across the USA that have programs established within the framework of the US EPA for water quality. New trends in trading programs range from northeastern programs established for the Long Island Sound (LIS) basin and the 2000 Chesapeake Bay Agreement that includes Maryland, Virginia, and Pennsylvania. Other trading programs are located in Midwestern Ohio and in the Pacific Northwest in Oregon.

The nutrient trading in the LIS basin addresses one of the northeastern region's important water quality problems, which has contributed to declining populations of fish and shellfish. The Connecticut Department of Environmental Protection (Con-DEP) established a Nitrogen Credit Exchange (NCE) that has helped in reducing N discharge to the LIS. Although the Con-Dep program does not include trading from the agricultural sector as of this writing, their successful program can serve as an example of some of the logistics and approaches used for N trading that may be applied to other regions of the United States.

The states of Maryland, Virginia, and Pennsylvania are required by new regulations to achieve significant reduction of nutrient and sediment flow from the Susquehanna and the Potomac watersheds, flow which is adversely affecting the Chesapeake Bay. The Pennsylvania Department of Environmental Protection (Penn-DEP) has instituted a nutrient trading program to encourage nonpoint sources to participate. The Penn-DEP certification procedures include a listing of credit-generating BMPs which can be submitted to the Penn-DEP for review (PADEP, 2006).

A large fraction of Ohio's rivers and streams do not meet state guidelines for fishing, swimming, and other designated uses. New guidelines require that wastewater treatment plants reduce the pollutions more aggressively than the current levels at the plants. Water treatment plants can generate economic savings by trading nitrogen credits on agricultural projects upstream of the plant, resulting in a greater reduction of nutrient loss at a significantly lower cost. Thus, the Water Conservation Sub-District (WCS) of the Miami Conservation District (MCD) is implementing the Water Quality Trading Program. The MCD funds proposals that reduce the most nutrient runoff to water bodies (http://www.miamiconservancy.org/WQTP/index.asp?data=dataXML.asp).

In the Willamette Basin in Oregon, temperature, bacteria, and mercury are three of its main pollutants (http://www.deq.state.or.us/wq/willamette/factsheets.htm). The Oregon Department of Environmental Quality (OR-DEQ) is encouraging point sources in need of upgrades to consider water quality trading. Instead of installing highly costly equipment, credits could be gained by implementing practices such as installing riparian buffers, which could save Oregon taxpayers millions of dollars and may also have some additional ecological benefits.

5.2. Air quality markets

Air quality markets are established programs that are intended to reduce the anthropogenic greenhouse gas emissions that contribute to Global Warming Potential. The Kyoto Protocol framework identified carbon dioxide (CO_2), methane (CH_4), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), sulfur hexafluoride (SF_6), and nitrous oxide as greenhouse gases. In North America, the only cap and trade system for these trace gases is the Chicago Climate Exchange (CCX) market (http://www.chicagoclimatex.com/). The NTT could quickly assess the potential savings in N_2O emissions (Delgado *et al.*, 2008c; Gross *et al.*, 2008). These N_2O savings estimated with the NTT may potentially be credited and traded in future markets for equivalents in carbon sequestration equivalents (Delgado *et al.*, 2008c; Lal *et al.*, 2009).

6. Summary and Conclusions

The use of N inputs in agricultural systems has heavily influenced the sustainability and economical viability of agricultural systems worldwide. These N inputs help maximize yields, which is necessary to supply food to the ever-growing world population. However, when more N than necessary is applied, the excess N applications result in increased N losses to the environment, which affects air and water quality. Recent developments in air and water trading markets may provide current and future opportunities for trading N savings.

The new N trading concept, a stand-alone NTT with GIS capabilities, and a new Internet prototype of an NTT were developed by the NRCS, in cooperation with the ARS-SPNR. Both the web-based and stand-alone prototypes allow users of this new technology to quickly determine how many potential N credits their farming operations can generate. These tools have straightforward, easy-to-use screens and users can conduct quick assessments of management practices.

Initial results suggest that these tools were capable of evaluating effects of best management practices and determining potential N savings to trade across a region of south central Colorado, Ohio, and Virginia. The GIS prototype will allow users to conduct quick assessments across a larger region, and to identify areas where losses are higher or where there will be greater potential to trade N savings. The NTT prototype is also capable of assessing the potential to trade in direct carbon sequestration equivalents due to savings from reductions of emissions of N_2O, and also indirect carbon sequestration equivalents due to savings in indirect N_2O losses. We suggest that such a tool could be used for air quality markets and for direct and indirect carbon sequestration equivalent markets. We propose in this chapter that the new NTT-GIS can be used to quickly identify the scenario that shows the greatest potential to maximize field-level savings in reactive N for environmental conservation, and to maximize N credits for potential trade of direct and indirect carbon sequestration equivalents.

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