SUSTAINING SOIL RESOURCES WHILE MANAGING NUTRIENTS

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Introduction

The focus of most nutrient management studies has logically been on economic viability and water quality. In this paper, we examine the wider issue of sustaining soil resources when developing practices designed to improve water quality. Sustainability when applied to crop production is often an emotionally charged word that has been used in many contexts. It has been used interchangeably with terms such as low-input sustainable agriculture, alternative agriculture, organic farming, regenerative farming, best management practices, and maximum economic yield (Keeney, 1990). Here we wish to use it in a more formal sense as defined by the 1987 Iowa Groundwater Protection Act. The Act defined sustainable agriculture as "the appropriate use of crop and livestock systems and agricultural inputs supporting those activities, which maintain economic and social viability whereas preserving the high productivity and quality of Iowa's land". Similarly, the National Food, Agriculture, Conservation, and Trade Act of 1990, Section 1603 defines sustainable agriculture as "an integrated system of plant and animal production practices that will, over the long term, enhance environmental quality and the natural resource base upon which the agricultural economy depends". Thus, to be sustainable in the Corn and Soybean Belt, a farming system not only needs to be economically viable and protect water quality but also must preserve or enhance the soil resource that makes the highly productive agriculture possible.

How do various management practices affect yield, water quality, and the soil resource? When discussing soil productivity we are primarily concerned with maintaining or building soil organic matter (SOM) within the topsoil, as it is SOM that provides much of the nutrient reservoir (fertility), determines physical characteristics that control infiltration, aeration, and aggregation associated with good soil tilth, and provides the energy or substrate for biological processes. SOM can be lost from a soil through two primary mechanisms – soil erosion and *in situ* decomposition. Soil erosion is a natural process, but enhanced erosion has been a consequence of agriculture from its inception. Sediment derived from soil erosion is the primary pollutant of surface waters today and a major cost to society. As it is the topsoil that erodes, these sediments are enriched in SOM and the nutrients required for crop production. Current soil conservation programs are targeted towards reducing soil erosion to the tolerable or "T" level as defined by the Revised Universal Soil Loss Equation and all management practices targeted for nutrient loss reduction must also keep soil loss below T. As topsoil protection from erosion is covered elsewhere in the Workshop, we will not examine it further here except to point out that achieving T alone is not sufficient for sustaining soil resources. Instead, we will concentrate on the second loss mechanism for SOM – decomposition.

SOM is composed of many different organic compounds ranging from fresh crop residues through their various decomposition products, to stable humus that is only very slowly decomposed to CO_2 and soluble compounds that can be leached from the soil. On average, soil humus contains about 5.6% N and 56% C for a C:N ratio of 10 (Waksman, 1938). Thus, we can speak interchangeably about either the soil organic carbon (SOC) pool or the total soil organic nitrogen (TN) pool when discussing SOM. To maintain SOC levels, the long-term input of C or

N into the humus pool must equal the long-term loss. Therefore, long-term gains and losses of either C or N from the organic soil fractions can be used to monitor changes in SOC.

Soil Nitrogen Mass Balance

Nitrogen mass balance calculations have been made at the field and watershed scales. At the field scale, studies by Karlen et al. (1998), Drinkwater et al. (1998), Jaynes et al. (2001), and Webb et al. (2004) have all shown agricultural systems with lower N inputs compared to N outputs. There have also been numerous watershed scale studies (David et al., 1997; Burkart and James, 1999; Goolsby et al., 1999; Libra et al., 2004), with studies using the more conservative net N input approach also finding a total N outputs exceeding total N inputs (McIsaac and Hu, 2004). All of these studies use the conservation of mass to compute the N balance, *i.e.*

$$\Delta \text{ inputs} - \Delta \text{ outputs} - \Delta \text{ soil residual mineral N} = \text{residual}$$

A residual > 0 indicates that inputs of N exceed losses from the field and N is available for other processes such as increasing SOM. A residual < 0 indicates that inputs do not balance outputs and that additional N must be coming from sources not included in the inputs such as decomposition of SOM to account for the observed losses. A residual = 0 indicates that N inputs and outputs from the field are in balance and therefore the production system is sustainable from a SOM perspective.

For a typical corn/soybean rotation, inputs of N to a field include the application of fertilizer and manure, N contained in rain and dry deposition, and N fixed by soybean. Outputs from the field include N removed with the grain harvest and NO_3 in deep drainage and runoff. Approaches for estimating each input and output are summarized below.

<u>Fertilizer and manure inputs</u> Fertilizer inputs are usually known and include N applied through sources such as anhydrous ammonia, urea-ammonium nitrate (UAN), ammonium nitrate, or ammonium sulfate and N associated with P fertilizers (e.g. mono-ammonium phosphate and diammonium phosphate). Manure inputs are based on total N at time of application minus a volatilization loss (Killorn and Lorimor, 2003) that depends on mode of application (i.e. injected, broadcast, etc).

<u>Wet and dry deposition</u> Nitrogen supplied by precipitation can be estimated from measurements made by the National Atmospheric Deposition Program (http://nadp.sws.uiuc.edu/). Across the Corn and Soybean Belt, average annual combined wet deposition of NO_3 and NH_4 ranges from 3.7 to 7.0 kg N ha^{-1} yr^{-1}. For dry deposition, the approximation used by Goolsby et al. (1999) can be used where dry deposition equals 0.7X of wet deposition.

<u>Fixation</u> Nitrogen fixation by soybean ranges considerably (McIsaac et al., 2002) not only because of the plant, soil, and climatic factors involved, but also because fixation depends on the availability of soil N to the plant (Russelle and Birr, 2004). Estimates of fixation also vary because of differences associated with methods (i.e. fertilizer replacement or isotopic (^{15}N) uptake) used to estimate its contribution. Barry et al., (1993) found a linear relationship between soybean grain yield (Mg ha^{-1}) and N fixed (kg ha^{-1}) as

$$N_{fixed} = 81.1 * \text{yield} - 98.5$$

This compares to a more conservative estimate when soybean yields are > 2.1 Mg ha^{-1} used by McIsaac et al. (2002)

$$N_{fixed} = 33.4 * \text{yield}$$

Grain removal The amount of N exported in grain depends on both the yield and protein content. Protein content for corn typically ranges from 60 to 90 g kg^{-1}, whereas in soybean it typically ranges from 336 to 375 g kg^{-1} (Russelle and Birr, 2004). Assuming a typical protein to N ratio of 6.25:1 (David et al., 1997), estimating the total N mass removed with corn or soybean is easily calculated. However, for other crops (e.g. wheat) the typical protein to N ratio would be 5.75 to 1.

Drainage Measuring N losses in percolation below the root zone is often very difficult. However, in fields where subsurface tile has been installed to improve drainage, the volume of flow and N losses in tile water can be directly measured and accounts for most of the percolation of N below the root zone.

Runoff Runoff losses of N can be measured at the edge of a field using flumes or other techniques to determine the volume of flow and from which samples can be collected to determine N concentrations. However, it is often safe to assume that very little N is lost in runoff, especially if injected or applied as a liquid (Gascho et al., 1998). Nitrate is very soluble in water and leaches below the soil surface at the start of each rainfall event. Therefore, it is generally not available for loss in surface runoff.

Weathering of the soil mineral fraction is not accounted for in this partial N balance, but this is generally considered to be trivial. Denitrification is not explicitly accounted for but this loss can be substantial in some locations and years. Unfortunately, it is extremely variable and difficult to estimate or measure accurately at the field scale (Parkin and Meisinger, 1989). Volatilization is also not accounted for, although such losses are generally minimal when N fertilizer is applied properly. Volatilization from manures is accounted for in the computation of N applied with manure inputs. N can also be lost directly from senescing plants (Francis et al., 1997), but the magnitude of this loss is variable and not well quantified for corn, soybean, or other crops. As presented, the partial N balance (Eq. 1) does not include these unaccounted for loss pathways, but it is important to note that all do occur in the field to some extent resulting in somewhat lower residual mass balances than computed in the next section.

Finally, the conversion of mineral N to organic N (immobilization) and the mineralization of N from SOM to mineral N are also not considered explicitly in the N mass balance. Instead this is captured by the residual term. A positive residual indicates surplus N that would be available to build additional SOM. A negative residual indicates that N is being supplied from an unaccounted for source, most likely mineralization or loss of SOM.

Examples

To illustrate how N management affects the sustainability of soil resources and crop production, we will examine data from three N management studies conducted in Iowa.

Deficit fertilization In a study described in Jaynes et al. (2001), three N fertilizer rates were replicated three times in a producer's field planted to a corn-soybean rotation. Nitrogen was applied in the spring after corn emergence at three multiples (1X, 2X, and 3X) of a base or target rate of 67 kg N ha^{-1}. Corn yields ranged from 6.63 to 10.73 Mg ha^{-1} over the 4-yr study with the economic optimum N rate being equal to about the 2X rate. Soybean yields were not affected by N application rate in the corn year and averaged 3.66 Mg ha^{-1}. By monitoring the tile drainage from each treatment plot, Jaynes et al. (2001) found that the annual flow-weighted NO_3 concentrations ranged from 11.4 mg L^{-1} for the 1X treatment to 18.8 mg L^{-1} for the 3X treatment. Using the MCL for NO_3 in drinking water, none of these N treatments could be characterized as sustainable from a water quality perspective, although lowering the N rate substantially lowered NO_3 concentrations in the drainage water.

To compute the N mass balance, inputs from the N and P-K fertilizations were measured. Inputs from wet deposition were estimated using measured precipitation and NADP average NO_3 and NH_4 concentrations in precipitation for central Iowa. Dry deposition was estimated to be 0.7X of wet deposition (Goolsby et al., 1999). Nitrogen fixed by soybean was estimated from the measured soybean yield and the relationship between yield and N fixed of Barry et al. (1993). Grain removal of N was determined using measured grain yield and protein content. Drainage losses of NO_3 were computed using measured drainage volume and NO_3 concentration in the water from tile drains installed 1.2 m below the surface. Runoff losses of N were not measured but considered negligible since the field was nearly level (< 1% slope) and runoff was observed only twice over 4 yr. Changes in soil mineral N were measured every fall within the top 1.2 m by collecting cores, extracting and analyzing for NO_3 and NH_4.

The 4-yr average partial N balance for each N rate in this study is shown in Table 1. Wet and dry deposition as well as N fixed by soybean was nearly identical for each treatment. N removed in grain harvest varied by about 15% due to treatment differences in corn yield. As shown for the NO_3 concentrations, the mass of N loss through drainage water was also a function of N rate, where the loss increased by approximately 64% as fertilization rates increased from 1X to the 3X. Changes in runoff and residual soil mineral N were nominal. Summing the inputs and outputs for these treatments shows residual values of -55, -26, and 47 kg ha^{-1} for the three treatments. Residuals of <0 for the 1X and 2X treatments means that more N was being lost from those systems than was being applied. This missing N had to come from some source unaccounted for in Table 1, with the most likely source being the soil organic N pool. The lower two N rates were thus effectively mining N from the SOM, which would result in a measurable decrease in SOM and a degradation of the soil resource over the long-term. Only for the 3X rate do we see a residual N balance > 0, indicating that more N was being applied than was being removed. Thus, only for the 3X treatment was SOM not being consumed, but rather sufficient N was being applied to potentially increase SOM. The existence of a positive N balance was also presumed responsible for a SOM increase, even with moldboard plowing, after 15 yr of continuous corn fertilized annually with approximately 200 kg N ha^{-1} on an Iowa Till Plain site (Karlen et al., 1998). SOM increases in that study accounted for approximately 42% of the N budget for the period. However, it is important to remember that Table 1 shows only a partial N balance and we are not considering additional N loss pathways such as denitrification or volatilization. Including these loss pathways would result in a smaller N balance than is shown.

In summary, whereas the economic optimum N fertilizer rate was approximately 134 kg N ha^{-1} for the 4-yr study, nitrate concentrations in the tile drainage water for all treatments exceeded the 10 mg L^{-1} MCL for drinking water and the lower N treatments (67 or 134 kg N ha^{-1}) were mining N from the SOM fraction. Thus, simply applying lower N fertilizer rates fails the definition of sustainability by not maintaining the long-term productivity of the soil whereas the high N fertilizer rate doesn't meet the definition because of high NO_3 concentrations leaving the field in tile drainage. Based on this assessment, the practice of deficit fertilization, although suggested as a viable alternative for solving NO_3 contamination of surface waters and the Northern gulf (Mitsch et al., 1999) is not a sustainable management practice with regard to long-term soil productivity.

Table 1. Partial N mass balance for 4-yr rate study by Jaynes et al. (2001).

	------------ N inputs ---------			----------------- N outputs -----------------				
Fertilizer rate	Total fertilizer applied	Total wet and dry deposition	Total fixed	Total grain removal	Total drainage loss	Total runoff†	Change in residual mineral N	N balance residual
	------------------------------ kg ha^{-1} ------------------------------							
1X	144	43	395	522	119	0	6	-55
2X	289	43	397	590	142	0	13	-26
3X	414	43	394	606	195	0	-7	47

†Not measured, but little runoff observed during 4-yr period.

<u>Cover crops and bioreactors</u> A second example evaluating the sustainability of alternative N management strategies comes from the unpublished data collected for a study reported by Jaynes et al. (2004) using cover crops and an in-field bioreactor (see Cooke et al., this Workshop). In their study, a corn/soybean rotation with conventional management and subsurface drainage was compared to the same rotation with an annual rye cover crop planted in the fall following each crop. In addition, a bioreactor consisting of wood chips buried in trenches on both sides of the subsurface drainage pipe was also investigated. The wood chips in the bioreactor served as a carbon source for denitrifying bacteria that reduced the nitrate in the shallow groundwater to N_2 before the nitrate could enter the subsurface drain and be carried from the field. Nitrogen fertilization for all treatments consisted of 224 kg ha^{-1} of N applied as UAN after corn emergence, which is on the upper end of the optimum N rate, but was used to stress the system with excess NO_3. Yields and grain protein were measured each year with corn averaging 11.8 Mg ha^{-1} and soybean averaging 2.77 Mg ha^{-1}. Tile drainage volume and nitrate concentration were monitored continuously. For the years 2000-2004, the average flow-weighted NO_3 concentration for the conventional treatment was 22.4 mg N L^{-1}, well above the MCL for drinking water. The flow-weighted average NO_3 concentration for the cover crop treatment was 14.4 mg N L^{-1}, although the average was below 10 mg L^{-1} in the three years where a cover crop was well established. The flow-weighted average NO_3 concentration for the bioreactor treatment was 8.5 mg N L^{-1}. Thus, the conventional treatment was not sustainable from a water quality perspective, nor was the cover crop treatment in every year, although it greatly reduced NO_3

losses in year where good cover crops could be established. The bioreactor treatment was sustainable from a water quality perspective as the NO_3 concentration in drainage was less than the MCL, but the longevity and profitability of this treatment remains to be determined.

A partial N balance for the conventional, cover crop, and bioreactor treatments is shown in Table 2. Again, runoff was minimal from these 0.4 ha plots and assumed to contribute little to N losses. As in the previous example, most of the N inputs were from inorganic fertilizer, although estimated fixation was a significant N source for soybean. Outputs were dominated by grain removal with tile drainage loss representing about a quarter of the N inputs. The overall N balance for the conventional system was slightly > 0, indicating the inputs and outputs were roughly in balance and SOM was not being mined from the soil. For the cover crop system, the N balance was substantially > 0, indicating a potential build up of soil N in the form of SOM most likely due to uptake of N and the increased biomass input from the rye cover crop. For the bioreactor, the partial N balance was >> 0, indicating a large N surplus. However, this surplus most likely did not represent a net gain of N within the soil but rather represented the increased denitrification that the bioreactor was designed to foster. Efforts to confirm the projected changes in SOM through direct measurements of SOM or more sensitive soil carbon fractionation are not planned for these plots for a few years because of the expected difficulty in detecting small changes in the large SOM pool.

The conventional production system in this case was sustainable from a soil productivity perspective, as the N mass balance indicates that the SOM content of the soil would be stable over the long term. However, this management system cannot be viewed as sustainable because of the high nitrate concentrations that leave the field in tile drainage. By adding a cover crop to the system, nitrate losses in drainage decreased substantially, but average NO_3 concentrations were still greater than the MCL. The cover crop also added biomass to the system that may combine with the nitrate that is not leached to form additional SOM, thus maintaining or enhancing long-term soil productivity. Based on these results, a corn/soybean rotation with a cover crop would be considered a sustainable system from both a water quality and soil quality perspective. Installing a bioreactor to the system dramatically reduces nitrate leaching and makes the system sustainable from a water quality perspective, but probably does little to enhance the soil resource.

Table 2. Partial N mass balance from 5-yr rate study by Jaynes et al. (2004).

	----------N Inputs -------------			-----------------N Outputs ---------------				
Treatment	Total N fertilizer applied	Total wet and dry deposition	Total fixed	Total grain removal	Total drainage loss	Total runoff†	Change in residual mineral N	N balance residual
	kg ha^{-1}							
Conventional	673	51	281	697	253	0	46	9
Cover crop	673	51	265	673	153	0	105	59
Bioreactor	673	51	258	676	100	0	88	118

†Not measured, but little runoff observed during 4-yr period.

Liquid manure Traditionally on diversified farms, animal manure was applied to provide essential plant nutrients and to build SOM by returning crop residues that had been used for bedding. However, as small- and medium-sized farms were replaced by concentrated animal feeding operations (CAFOs) and separate crop production enterprises, animal manure became to be considered more of a waste than a resource. Developing systems that reverse this perception and show that manure can be utilized in environmentally sound and economically profitable ways has been a research focus since the early 1990s (Hatfield and Stewart, 1998). This transition was not without many challenges associated with all aspects of the animal, crop, manure, and soil management systems (Karlen et al., 2004). Changes in manure management resulting in less solid (bedding) material, variability in nutrient composition as storage facilities were emptied, limited time for application, and the difficulty of regulating application rates are just a few examples.

With regard to sustainability of the soil resources and the potential impact on water quality, a 6-yr study conducted on Iowa Till Plain soils near Nashua, IA using liquid swine manure as the N source provided the following insights. Tile drainage volume was highly variable among the 0.4 ha (1 acre) plots, presumably because of subtle differences in slope and inherent soil characteristics. This variation in drainage volume in addition to variation in seasonal precipitation, current year and prior manure application rates (caused by variation in nutrient composition and application challenges), and the crop (corn or soybean) being grown resulted in NO_3 losses that varied from 4 to 48 kg N ha^{-1} yr^{-1} during the 6-yr study (Karlen et al., 2004). When averaged for continuous corn, drainage loss accounted for 16% of the applied N, whereas for the corn-soybean rotation it accounted for only 10%. Grain yield was also variable averaging 6.4 Mg ha^{-1} for continuous corn (range 2.8 to 8.4 Mg ha^{-1}), and 7.9 for corn (range 5.5 to 9.8 Mg ha^{-1}) and 3.4 Mg ha^{-1} for soybean (range 2.6 to 3.9 Mg ha^{-1}) in the 2-yr corn-soybean rotation, respectively. The measured amount of N removed with the grain crops averaged 85, 100, and 182 kg ha^{-1} for the continuous corn and the corn and soybean phases of the rotation. Summing inputs and outputs for this manure study shows a substantial residual N balance for continuous corn (Table 3), but the combined corn/soybean rotation residual was negative. Thus, continuous corn may be building SOM in this field whereas the corn/soybean rotation was probably mining N from the SOM. However, measurements of SOM in the surface 20 cm (Karlen et al. 2004) did not reveal any significant changes over the 6 yr of the study. Perhaps measuring one of the more active C/N pool such as particulate organic matter (Cambardella and Elliot, 1993) would be more sensitive to changes in SOM content.

Table 3. Partial N mass balance for a 6-yr swine manure study by Karlen et al. (2004).

	----------N Inputs -------------			-----------------N Outputs ---------------				
Treatment	Total N[§] applied	Total wet and dry deposition	Total fixed	Total grain removal	Total drainage loss	Total runoff[†]	Change[¶] in residual mineral N	N balance residual
	---------- kg ha^{-1} ----------							
Continuous corn	958	63	0	510	156	0	82	273
Corn phase[‡]	794	63	0	600	84	0	116	57
Soybean phase[‡]	0	63	1058	1092	150	0	2	-123

[†]Not measured, but very little runoff was observed during the 6-yr study.
[‡]Both phases of a corn-soybean rotation were present each year

§A 2% loss from volatilization was assumed for liquid injection.
¶ Estimated 6-yr values based on 1996, 1997 and 1998 measurements (Bakhsh et al., 2001)

Limitations

As illustrated by our inability to detect SOM differences in soil samples from the three studies described above, assessing the impact of nutrient practices on SOM content and long-term productivity of soil is difficult because we are trying to measure small changes in a large quantity. For example, Russell et al. (2005) were measured lower particulate organic carbon content in a long term corn – soybean rotation compared to continuous corn and a corn – corn – oat – alfalfa rotation, but could only detect an increase in SOM with increasing fertilizer rates for a continuous corn rotation. In contrast, Omay et al. (1997) were able to measure a 32% increase in potentially mineralizable C due to application of N fertilization, but could not consistently measure differences due to crop rotation. Much of the inability to quantify differences is due to the large variability in measured organic fractions in soil.

Therefore we have substituted an N mass balance for a C mass balance because N inputs and losses are more easily measured. Nevertheless, there are large uncertainties in N mass balance computations even at the field scale. On the input side, the contribution of soybean to the available N pool through fixation is highly uncertain and variable (Russelle and Birr, 2004). Whereas more N is removed in grain harvest of soybean than is fixed (Heichel, 1987), soybean still fixes considerable amounts of N. Better quantification of this fixed N would greatly reduce the uncertainty of the input calculations.

On the output side, volatilization of N from fertilizer, manure, and soil are important but also difficult to quantify as they are weather and practice specific. Estimates of the N emissions from senescing plants vary over an order of magnitude but have been rarely measured. New methods need to be developed for measuring N losses at the field-scale from senescing plants. Denitrification is highly variable over time and space (Parkin and Meisinger, 1989) making annual field-wide estimates suspect.

Finally, direct measurement of changes in SOM that could detect changes in a few years would greatly ease the assessment of the sustainability of potential practices. New techniques need to be developed to allow measurement of changes in the large SOM pool in most soils from the Corn and Soybean Belt.

Summary

Effects of nutrient management practices need to be evaluated against not only economics and water quality, but also against long-term soil productivity to ensure a profitable and environmentally sustainable agricultural production system within the Corn and Soybean Belt. Soil organic matter is an important indicator of soil productivity and soil tilth as many of the biological, chemical, and physical properties of a soil that are important for crop production in the Corn and Soybean Belt are strongly influenced by SOM levels. For the studies reviewed here, the negative N mass balances for the 1X treatment in example 1 and the corn/soybean rotation under liquid manure application in example 3, are equivalent to ~ 0.2% yr^{-1} loss in SOM from the top 20 cm of the profile. Conversely, the positive N balances for the cover crop in Table

2 and the continuous corn with liquid manure application in Table 3, represent increases in SOM of ~ 0.2% and 0.9 % yr^{-1}, respectively. While small in terms of our current ability to directly measure, these changes represent about a 5% loss in SOM over 30 yr for the first two, and a gain of 5% and 26% in 30 yr for the latter two. Compared to the 10-20% minimum differences in SOM required to detect significant differences typically found in soil carbon studies (Russell et al., 2005), it is of little wonder that losses in SOM due to organic matter decomposition has been difficult to quantify with direct SOM measurements

Nutrient management practices need to be assessed for their ability to enhance or maintain SOM content in addition to their impact on yield and profit. Just as nutrient management studies are incomplete if they consider only yield and ignore water quality, water quality studies evaluating nutrient practices that neglect the long-term effects on the soil resource are also incomplete. Future nutrient management studies must be designed to measure impacts on soil and water resources as well the economics of various practices.

Acknowledgements
The authors wish to thank the contributions of the following individuals who served as Panelists at the Workshop and helped review and provided suggestions on all drafts of this paper. William Crumpton, Iowa State University, Ames, IA; Lowell Gentry, Michigan State University, East Lansing, MI; and Michael Russelle, USDA-ARS, St. Paul, MN.

References
Bakhsh, A., R. S. Kanwar, D. L. Karlen, C. A. Cambardella, T. B. Bailey, T. B. Moorman, and T. S. Colvin. 2001. N-management and crop rotation effects on yield and residual soil nitrate levels. Soil Sci. 166(8):530-538.

Barry, D.A.J., D. Goorahoo, and M.J. Goss. 1993. Estimation of nitrate concentrations in groundwater using a whole farm nitrogen budget. J. Environ. Qual. 22:767-775.

Burkart, M.R., and D.E. James. 1999. Agricultural-nitrogen contributions to hypoxia in the Gulf of Mexico. J. Environ. Qual. 28:850-859.

Cambardella, C.A., and E.T. Elliott. 1993. Carbon and nitrogen distribution in aggregates from cultivated and native grassland soils. Soil Sci. Soc. Am. J. 57:1071-1076.

David, M.B., L.E. Gentry, D.A. Kovacic, and K.M. Smith. 1997. Nitrogen balance in and export from an agricultural watershed. J. Environ. Qual. 26:1038-1048.

Drinkwater, L.E., P. Wagoner, and M. Sarrantonio. 1998. Legume-based cropping systems have reduced carbon and nitrogen losses. Nature 396(19):262-265.

Francis, D.D., J.S. Schepers, and A.L. Sims. 1997. Ammonia exchange from corn foliage during reproductive growth. Agron. J. 89:941-946.

Gascho, G.J., R.D. Wauchope, J.G. Davis C.C. Truman, C.C. Dowler, J.E. Hook, H.R. Sumner, and A.W. Johnson. Nitrate-nitrogen, soluble, and bioavailable phosphorus runoff from simulated rainfall after fertilizer application. Soil Sci. Soc. Am. J. 62:1711-1718.

Goolsby, D.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, R.P. Hooper, D.R. Keeney, and G.J. Stensland. 1999. Flux and sources of nutrients in the Mississippi-Atchafalaya River basin. Topic 3. Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico. NOAA Coastal Ocean Program Decision Analysis Series No. 17. NOAA Coastal Ocean Program, Silver Springs, MD.

Hatfield, J.L., and B.A. Stewart (eds.) 1998. Animal Waste Utilization: Effective use of manure as a soil resource. Sleeping Bear Press, Chelsea, MI. 320 p.

Heichel, G.H. 1987. Legume nitrogen: Symbiotic fixation and recovery by subsequent crops. p. 63-80. *In*: Z.R. Helsel (ed.) Energy in Plant Nutrition and Pest Control. Elsevier, Amsterdam.

Jaynes, D.B., T.S. Colvin, D.L. Karlen, C.A. Cambardella, and D.W. Meek. 2001. Nitrate losses in subsurface drainage as affected by nitrogen fertilizer rate. J. Environ. Qual. 30:1305-1314.

Jaynes, D.B., Kaspar, T.C., Moorman, T.B., and Parkin, T.B. 2004. Potential methods for reducing nitrate losses in artificially drained fields. p. 59-67. Internat. Drainage Symp. Sacramento, CA. Mar. 21-24, 2004.

Karlen, D.L., Ajay Kumar, R.S. Kanwar, C. A. Cambardella, and T.S. Colvin. 1998. Tillage system effects on 15-year carbon-based and simulated N budgets in a tile-drained Iowa field. Soil & Till. Res. 48:155-165.

Karlen, D.L., C.A. Cambardella, and R.S. Kanwar. 2004. Challenges of managing liquid swine manure. Appl. Eng. Agric. 20(5):693-699.

Keeney, D. 1990. Sustainable agriculture: Definition and concepts. J. Prod. Agric. 3:281-285.

Killorn, R., and J. Lorimor. 2003. Managing manure nutrients for crop production. PM 1811. Iowa State Univ. Extension. Ames, IA.

Libra, R.D., C.F. Wolter, and R.J. Langel. 2004. Nitrogen and phosphorus budgets for Iowa and Iowa watersheds. Iowa Dept. Natural Resour., Iowa Geological Surv. Tech. Info. Ser. No. 47.

McIsaac, G.R., and X Hu. 2004. Net N input and riverine N expert from Illinois agricultural watersheds with and without extensive tile drainage. Biogeochem. 70:251-271.

McIsaac, G.R., M.B. David, G.Z. Gertner, and D.A. Goolsby. 2002. Relating net nitrogen input in the Mississippi river basin to nitrate flux in the lower Mississippi river: A comparison of approaches. J. Environ. Qual. 31:1610-1622.

Mitsch, W.J. J.W. Day Jr., J.W. Gilliam, P.M. Groffman, D.L. Hey, G.W. Randall, and N. Wang. 1999. Reducing nutrient loads, especially nitrate-nitrogen to surface water, groundwater, and the Gulf of Mexico. Topic 5. Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico. NOAA Coastal Ocean Program Decision Analysis Series No. 19. NOAA Coastal Ocean Program, Silver Springs, MD.

Omay, A.B., C.W. Rice, L.D. Maddux, and W.B. Gordon. 1997. Changes in soil microbial and chemical properties under long-term crop rotation and fertilization. Soil Sci. Soc. Am J. 61:1672-1678.

Parkin, T.B., and J. Meisinger. 1989. Denitrification below the crop rooting zone as influenced by surface tillage. J. Environ. Qual. 18:12-16.

Russell, A.E., D.A. Laird, T.B. Parkin, and A.P. Mallarino. 2005. Impact of nitrogen fertilization and cropping system on carbon sequestration in Midwestern mollisols. Soil Sci. Soc. Am. J. 69:413-422.

Waksman, S.A. 1938. Humus. Williams and Wilkins Co., New York.

Webb, J. S. Ellis, R. Harrison, and R. Thorman. 2004. Measurement of N fluxes and soil N in two arable soils in the UK. Plant and Soil 260:253-270.