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Quantifying Cradle-to-Farm Gate Life-Cycle Impacts Associated with Fertilizer Used for Corn, Soybean, and Stover Production

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Table of Contents

List of Tables

List of Figures

Executive Summary

Fertilizers used to increase the yield of crops used for food or bio-based products can migrate through the environment and potentially cause adverse environmental impacts. Nitrogen fertilizers have a complex biogeochemical cycle. Through their transformations and partitioning among environmental compartments, they can contribute to eutrophication of surface waters at local and regional scales, groundwater degradation, acid rain, and climate change. Phosphate fertilizers have a simpler fate in the environment, although leaching of soluble and bound phosphorus is an important contributor to eutrophication.

Eutrophication is considered one of the most pervasive problems affecting water quality in the United States, especially in the Midwest where fertilizers are used extensively for agriculture. In the process of eutrophication, the presence of excess N and P nutrients allows over production of plant biomass in waterways. The eventual degradation of this biomass consumes oxygen resulting in hypoxic conditions (low oxygen concentrations) in the most severe cases of eutrophication. Fertilizer use on corn and soybean farms in the Midwest is considered one of the primary contributors to the growing hypoxic zone in the Gulf of Mexico. Through a combination of excessive nutrient loads and hydrodynamic conditions, a region along the coast of Louisiana that is approximately the size of the State of Massachusetts is considered ecologically dead most summers. This results in the death of species that are not sufficiently mobile and changes in biodiversity and food webs throughout the region as larger species migrate to other locations. Researchers for federal agencies have suggested that reducing the average nitrogen load by 30% will help to limit the hypoxic zone to acceptable levels. Other researchers predict that a 40-45% reduction in TN would be required to meet this goal.

With an increase interest in the use of corn, soybeans and corn stover for bio-based products and fuels, it is important to understand the relative environmental benefits and deleterious impacts associated with this growing market. A team of researchers lead by NREL completed a life cycle assessment (LCA) for stover harvest and conversion to ethanol for transportation fuels. Their report focused on the green house gas emission benefits associated with biofuels as balanced by potential detriments to soil health (carbon content and erosion). Eutrophication was identified as an important issue by stakeholders involved with this project, but limited resources prevented this environmental impact category from being addressed. Thus, the goal of the work presented here was primarily to fill that gap, thereby providing a more complete picture of the overall environmental impacts associated with bio-based products.

The nutrient leaching model developed here was coupled with LCA data describing emission occurring during fertilizer manufacture and energy production and consumption, providing a cradle-to-farm gate life cycle inventory (LCI) for corn, soybeans and stover. Three separate scenarios were considered:

- Scenario 1: the base case, considered corn-soybean rotations (C-S) with conventional till and no stover collection;
- Scenario 2: C-S with no till and stover collection at a maximum rate allowable with acceptable erosion levels; and,
- Scenario 3: the same as 2 except for continuous corn (C-C) rather than C-S.

The LCI for each of these scenarios was quantified and used to determine eutrophication, acidification and global warming potentials for the three scenarios. Eutrophication was calculated for each year in a 13-year study period to incorporate variability with rainfall. Other impacts were only calculated as averages.

The total nitrogen (TN) and total phosphorus (TP) leaching models that correlate the fraction of fertilizer nitrogen (FN) and fertilizer phosphorus (FP) that leaches with annual rainfall in each county provides a good representation of the measured variability in nutrient loads discharged from eastern Iowa watersheds to the Mississippi River (Figures ES1 and ES2). The ability to calibrate this model reduces the

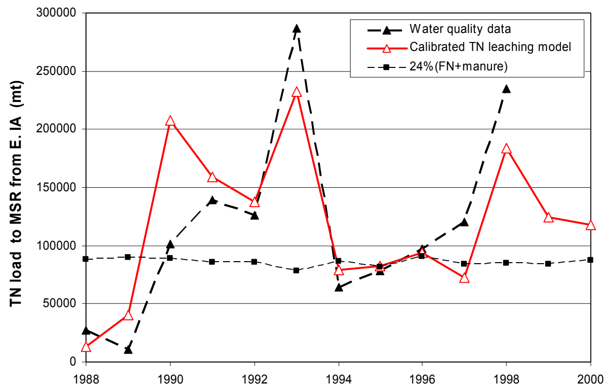


Fig. ES-1: Calibrated TN leaching model compared with actual measured TN loads to the Mississippi River

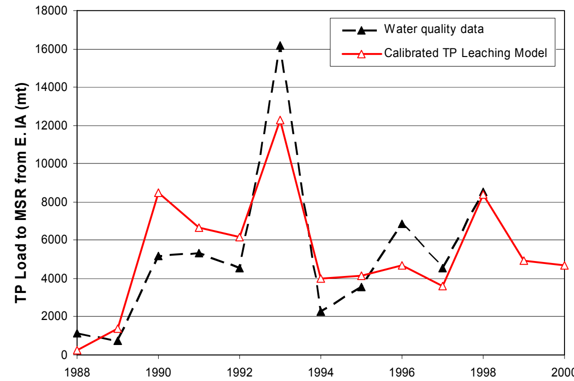


Fig. ES-2: Calibrated TP leaching model compared with actual measured loads to the Mississippi River

uncertainty in the leaching estimates for the base case scenario more than would be possible without the site-specific water quality data. The calibrated leaching model (solid line in Figs. ES-1 and ES-2) captures the wide variability in the annual TN and TP loads to the Mississippi River from the eastern Iowa watersheds. In contrast, a standard method for estimating nutrient leaching in other lifecycle assessments based on a constant fraction of the nutrients leached (▪ symbols in Fig. ES-1), provides a reasonable average, but does not capture the annual variability.

There are significant uncertainties associated with the allocating these total nutrient loads among the various crops planted in rotation. The allocation process is an important step in an LCA to define of how much of the total environmental emissions can be attributed to corn versus soybean versus stover products. Two radically different approaches were used to estimate the allocation of nitrogen flows between corn and soybeans. Estimates of the TN leached from the overall C-S system that can be allocated to corn range from 60 and 99%, with the balance allocated to soybeans. This difference stems from a poorly understood symbiosis of nitrogen flows within the C-S rotation that is difficult to integrate into an LCA.

The primary focus of this research was the eutrophication potential. The eutrophication potentials presented here include only nutrient emissions from the agricultural activities described in the three scenarios. Other agricultural inputs (e.g., animal feed lots) and point sources were not included in these totals. TN and TP flows to the Mississippi River and Gulf of Mexico were aggregated into an overall eutrophication potential, which is expressed as an equivalent mass of NO_3-N. The eutrophication loads predicted through the nutrient flow model developed here were compared to the target 30% reduction in TN flows required to limit the size of the Gulf of Mexico hypoxic zone and recommended water quality standards to control eutrophication at a local level.

The results of this analysis show that the eutrophication potential for the base case (scenario 1) already exceeds acceptable limit (solid line, Fig. ES-3). TN and TP discharges from C-S lands also exceed the maximum loads defined by the proposed water quality standards in each of the 13

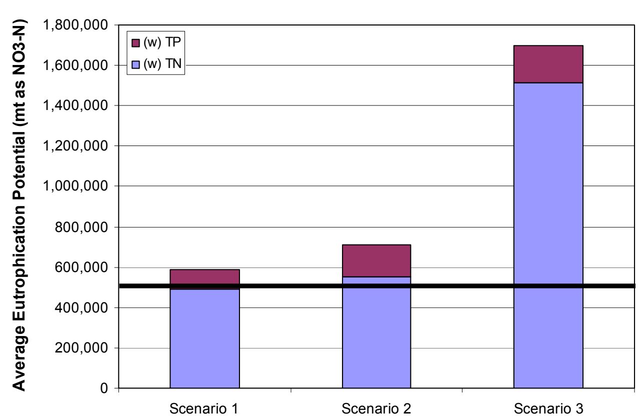


Fig. ES-3: Average eutrophication potential for the three scenarios (solid line represents goal for limited hypoxia; 30% reduction of ’80–‘96 TN average) years of this study. Limits established by the goal of a 30% reduction in the average TN load are also exceeded in approximately half of the years. If additional sources of nutrients discharged from the study area are considered, it is estimated that the eutrophication potential is higher than recommended except under drought conditions.

Changing the current C-S rotation to also harvest stover for biofuel production increases the eutrophication potential (scenario 2; Figure ES-3). With the assumptions used in this analysis, the C-S-stover scenario (2) results in a 21% increase in the eutrophication potential, while the C-C-stover scenario (3) almost tripled the total load of TN and TP (as equivalent NO3-N). In either case, the increased load is to a system that has already exceeded its assimilative capacity for nutrients. For the C-S system, it is likely that careful management of fertilizers used at the farm could help to limit nutrient leaching. With the very high nitrogen demand in a C-C system, however, it is not likely that management practices could be sufficient to overcome the detrimental effects of eutrophication resulting from the high leaching rates.

The high nitrogen use in the C-C scenario also increases the use of fossil fuels for fertilizer production and associated emissions of species contributing to acid rain and global warming (GWP) potentials. The increase in acidification potential (+6% over the base case) is attributed to increases in NO production in soils with increased FN use and increased NO_x from fossil fuel consumed to generate the increased energy necessary for fertilizer manufacture. Increases in the global warming potential (+71% over the base case, not including benefits of carbon sequestered in soil or crop) are attributed to methane emissions from natural gas used in FN manufacture and N_2O emissions from nitrification and denitrification of the additional FN.

Scenario 2 (C-S) actually reduces the global warming potential relative to the base case (-3%) and has essentially no impact on acidification. The reduction in the GWP is related to the reduced level of soil mineralization with no till and the resulting reduction in N_2O emissions.

There is some uncertainty in the quantitative results presented here due to assumptions used throughout the model to describe fertilizer use and their fate. The uncertainties in the life cycle and impact results are the least for scenario 1, for which site specific data and mass balance checks were used to verify the model as much as possible. There are greater uncertainties with scenario 2 and 3. Assumptions required to quantify differences in fertilizer needs and leaching under no-till practices were required. There has not yet been any analysis of the sensitivity of the conclusions to the uncertainty in these parameter values.

Based on the results presented thus far and the acknowledged limitations of the model – the following recommendations for continued analyses can be made:

- Consider stover only from C-S rotations in further analyses.
- Improve the erosion modeling to incorporate variability with rainfall. This could help to improve leaching models by quantifying changes in sediment-bound nutrient loads in a no till versus conventional till system.
- Use the framework developed here that utilizes water quality data to generate leaching models for herbicides. LCI data for herbicide manufacture also need to be developed.
- Utilize additional resources to improve predictions of nutrient fate in a no till versus conventional till system and perform a sensitivity analysis on these processes.
- Compare results to related studies and guidelines currently used to define the acceptability of bio-based products, especially for federal procurement.

The generation of this life-cycle data was required to provide a more comprehensive understanding of the environmental impacts associated with increased use of biomass for fuels and other products. The LCA quantifies impacts on very different components of the environment, but does not judge which of these components are more important. The LCA results are necessary, but not sufficient to allow decisions regarding future energy sources to be made. In the United States, the goal of current policies is to reduce our dependence on imported energy. The LCA results can be used to quantify the environmental benefits and detriments associated with this goal and can help identify key concerns that should be addressed as this project is continued (e.g., improved nutrient management to reduce water quality degradation). At some point, a balance needs to be defined between the various goals of different interest groups in a manner that then overall environmental impact from biomass fuels and products is considered acceptable, leading to a sustainable materials and energy source for the future.

1. Introduction

Consequences of Increased Agricultural Activity for Biomass Fuel and Products

Corn, soybeans and corn stover are all valuable feedstocks for conversion of biomass into consumer goods. Utilizing these agricultural products and residues creates a potential for both environmental benefits and deleterious impacts. The national use of these products could have a disproportionate negative impact in the Midwestern states on the soil and water resources, while having positive impacts on air quality and global climate change over a wider geographic scale. Many studies completed to date that have quantified the environmental impacts of bio-based products have focused on the air quality and greenhouse gas (GHG) benefits (Wang, 1999; Sheehan et al., 2002 Heller et al., 2003). There are clear benefits to using bio-based materials, especially in terms of greenhouse gas generation. Plant growth consumes atmospheric carbon dioxide is transformed to plant matter. Eventually, the carbon is released back to the environment at the end-of-life stage of a bio-based product or fuel. However, that release results in a near zero net GHG emission. In comparison, combustion of fossil fuels cause carbon sequestered in the subsurface for millennia to be added to our atmospheric carbon dioxide load.

Studies have not been completed that adequately evaluate the balance between these benefits - GHG and other air emissions - and potential soil and water quality degradation. Sheehan et al. (2002) completed a life cycle assessment (LCA) designed to evaluate the environmental impacts of corn stover harvest in Iowa and subsequent conversion to fuel-grade ethanol. Their study included a thorough analysis of the agricultural activity on the long-term sustainability of soil health, as defined by erosion and soil carbon concentrations. They found that a fraction of the stover can indeed be harvested while maintaining tolerable levels of soil erosion and steady-state concentrations of soil carbon over a long-term. Thus, the GHG and air emission benefits of using stover for ethanol production far outweigh problems associated with soil degradation.

The application of nutrients to croplands is critically important for improving crop yields and productivity of farmland. Only approximately 50% of the applied nutrients are integrated into plant mass, however. The remaining nutrients accumulate in soil, or are emitted to the atmosphere (NO, NH_3, N_2O) or water bodies as soluble components (NO_3, PO_4) or as a component of soil that is eroded. Agricultural activities are a significant contributor to the substantial increase of both reactive nitrogen and phosphorus (Smil, 2000; Galloway, 1998) in our environment. These increases have contributed to the degradation of both air and water resources. Life cycle studies on agricultural processes and products that integrate all of these flows are very limited. Nutrients used on cropland are non-point sources of pollution to water bodies. This necessarily makes it more difficult to quantify these flows in a quantitative life cycle inventory.

Through stakeholder meetings, the importance of nutrification – or eutrophication - of surface waters was recognized as a critical impact for the stover LCA (Sheehan et al., 2002). Resource limitations prevented these lower priority issues from being completed as part of the original LCA. A general goal of this report is to fill that void.

General Problems Associated with Nutrient Pollution

Nutrient enrichment frequently ranks as the top cause of impairment to our Nation's water resources (USEPA, 2000a), especially in the Midwest where agricultural practices increase the rate of anthropogenic nitrogen and phosphorus in the environment. These non-point sources create far higher loads to surface and groundwater bodies than most point sources (e.g., wastewater treatment plants). Nutrients in water systems can cause problems ranging from aesthetic impacts to serious human health problems.

While some nitrogen and phosphorus is required in all surface water bodies as nutrients, too much nitrogen can cause excess biological productivity, which leads to other problems (Figure 1). Algal blooms, for example, choke out other species and reduce light penetration into the water body causing underwater vegetation to die and altering habitat. As the algae die and sink, the biological oxygen demand that they impart can significantly reduce oxygen concentrations, thereby creating conditions that interfere with the recreational use of lakes and estuaries, increase costs to treat the water to drinking water quality standards, and the decrease the health and diversity of indigenous fish, plant, and animal populations. This condition is termed eutrophication.

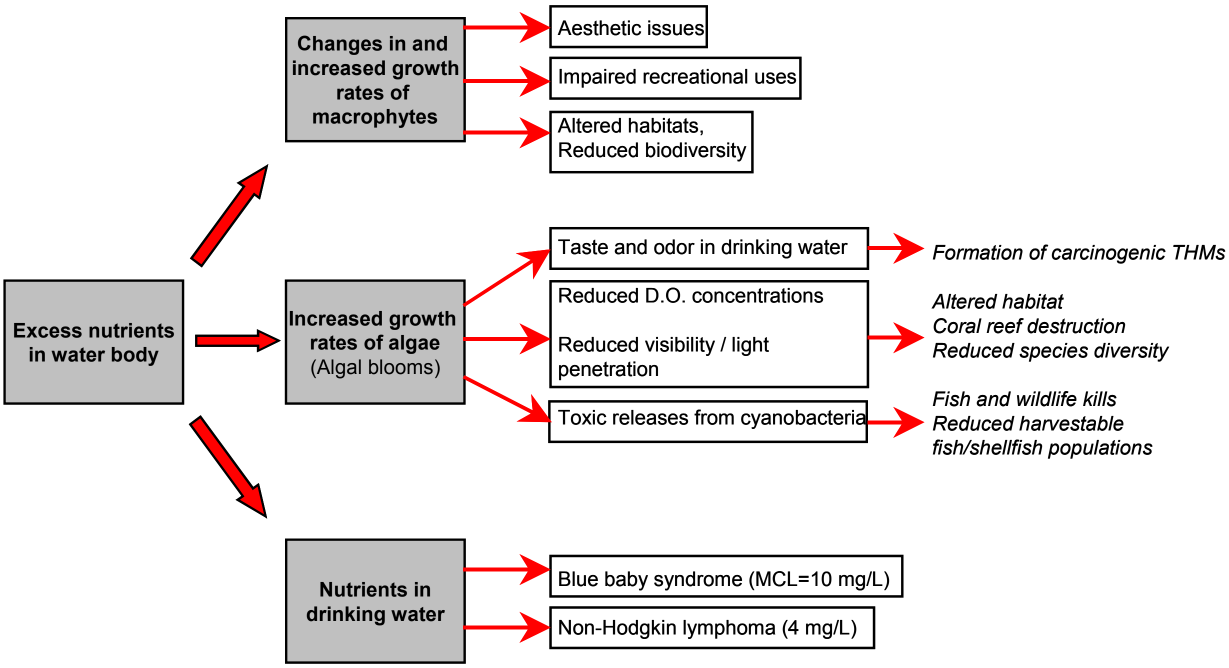


Figure 1. General impacts of excess nutrients in our water supplies
(adapted from Carpenter, 1998, US EPA, 2000a)

Low levels of oxygen in the Gulf of Mexico near the mouth of the Mississippi River are a form of severe eutrophication. Large areas of the Gulf, often similar in size to the State of Massachusetts, are considered ecologically dead. Mobile species leave this area, while smaller and immobile species succumb to the inhabitable low oxygen concentrations. This hypoxic zone was initially observed in the mid 1980s (Turner and Rablais, 1994) and has been the subject of numerous studies since (e.g., Scavia et al., 2003). The use of large quantities of inorganic fertilizers for row crop production (mostly soybeans and corn) and animal husbandry operations in the Midwest have contributed substantially to the nutrient load delivered to the Gulf of Mexico and the subsequent eutrophication processes that cause hypoxia.

Objectives of Study

The goal of this research is to understand and quantify the impacts that the increased utilization of dedicated biomass crops (corn, soybeans) and agricultural residues (corn stover) for bio-based energy and products will have on our Nation’s water resources. This research extends the stover LCA work already completed (Sheehan et al., 2002) by incorporating water resources impacts, especially focusing on nutrient use during the feedstock side of the LC. Specific objectives of the overall research project include:

1. Quantify the loads of agrichemicals to ground and surface water bodies expected from corn-soybean rotations and continuous corn, with and without stover collection, on eastern Iowa farms.
2. Identify and quantify the most suitable impact categories, characterization and normalization methods to incorporate these water quality impacts into an LCA
3. Determine the LCA water resources impacts for a site specific watershed-scale case for which existing water quality data are available

The report focuses on the use of nutrient agrichemicals only. By completing this work first, a framework has been established to extend the study to other agrichemicals and/or crops.

The stover LCA (Sheehan et al., 2002) was developed for the State of Iowa using county and state average data for agricultural practices and soil characteristics. For consistency, the work presented here also focuses on Iowa. A smaller subset of Iowa counties is considered here, however, to include a set of specific watersheds in eastern Iowa. These eastern Iowa watersheds have similar geologic and hydrologic properties. Additional details about this region are provided later.

2. Background

Nutrient Use for Corn-Soybean Production

Agricultural practices in the Midwestern United States rely heavily on chemicals to boost the productivity of the farmland. Inorganic nitrogen fertilizer (FN) use in the United States has increased from 10 million tons per year in the mid-1970s to over 45 million tons per year in the mid 1990s, with an average of 102 kg N/ha cropland applied in 1998 (Burkart and Stoner, 2002). Animal manures also contribute to the nitrogen load to the environment. In North America, however, manure application rates (kg/ha cropland) in 1999 were less than 50% of the inorganic fertilizer application rates (Burkart and Stoner, 2002) and less than 20% of corn fields receive organic fertilizer. These increases in FN are a significant contributor to the recent tripling of reactive nitrogen in the global environment (Galloway et al., 2003)

Burkart and Stoner (2002) identified 9 separate agricultural regions within the United States. The "corn, soybeans, hogs" category has by far the highest rate of nitrogen fertilizer application rates (median = 60 kg/ha, n=605). In Iowa, application of nitrogen fertilizers to corn were much higher than the average for this region (Figure 2). Nitrogen fertilizer is used on almost all cornfields and approximately 10% of soy fields. The rate of fertilizer use has declined somewhat on cornfields since the high levels used in the 1980s when the environmental and economic costs of over fertilizing became apparent. FN use on soy, however, is quite variable and at its highest level in the year 2000.

Like nitrogen, global mobilization of reactive P in the environment has roughly tripled compared to natural flows (Smil, 2000). The application of inorganic phosphorus fertilizers (FP) for crops is a substantial contributor to this overall increase in P flows. Global consumption of FP reached approximately 5 Mt P/yr in 1960 and increased to a peak of 16.5Mt/yr in 1988. Total consumption has since dropped and leveled at approximately 12Mt/yr (Smil, 2000)

Substantially lower rates of FP and FK are used compared with FN, especially on corn. However, FP application rates still generally exceed the rate at which P is removed from agricultural systems in the form of grain, leading to a long term increase in soil P concentrations (Sharpley et al., 2003). In 2000, 61% of the soil samples collected in Iowa near P-sensitive waters had extractable P concentrations greater than optimal concentrations required for crop growth. This percentage was up from 49% in 1997, ranking Iowa among the states with the highest percent of samples with P concentrations greater than optimum, and among the states with the greatest increase in that percentage between 1997 and 2000 (Sharpley et al., 2003). Figure 2 quantifies the rates of FP application in Iowa for soy and corn.

Environmental Fate and Impacts Associated with Nutrient Flows

Nitrogen cycling and fate in the environment

As the primary component in air, nitrogen is transformed into many different forms that are required for life. Industrial, physical (lightening) and biological (e.g., legumes) processes transform non-reactive nitrogen (N_2) into reactive forms, among them, ammonia and/or nitrate that can be used directly in living systems (US EPA 2000a, Galloway et al., 2003). These living systems then release mineralized forms of nitrogen (ammonia in urea, proteins etc.) as waste products or during decomposition after death (Galloway et al., 2003). Soil bacteria convert ammonium compounds into nitrate (nitrification). Under anaerobic conditions and in the presence or organic carbon, denitrifying bacteria transform nitrate into N_2, with nitrous oxides (NO, N_2O) as intermediate products that can be released to the atmosphere. Although this denitrification process is often considered to be an undesirable loss in terms of N available for plant growth, it is one of the few natural processes that convert reactive nitrogen back to its non-reactive state (N_2) (Galloway et al., 2003). Chemical reactions involved in these processes are summarized in Table 1.

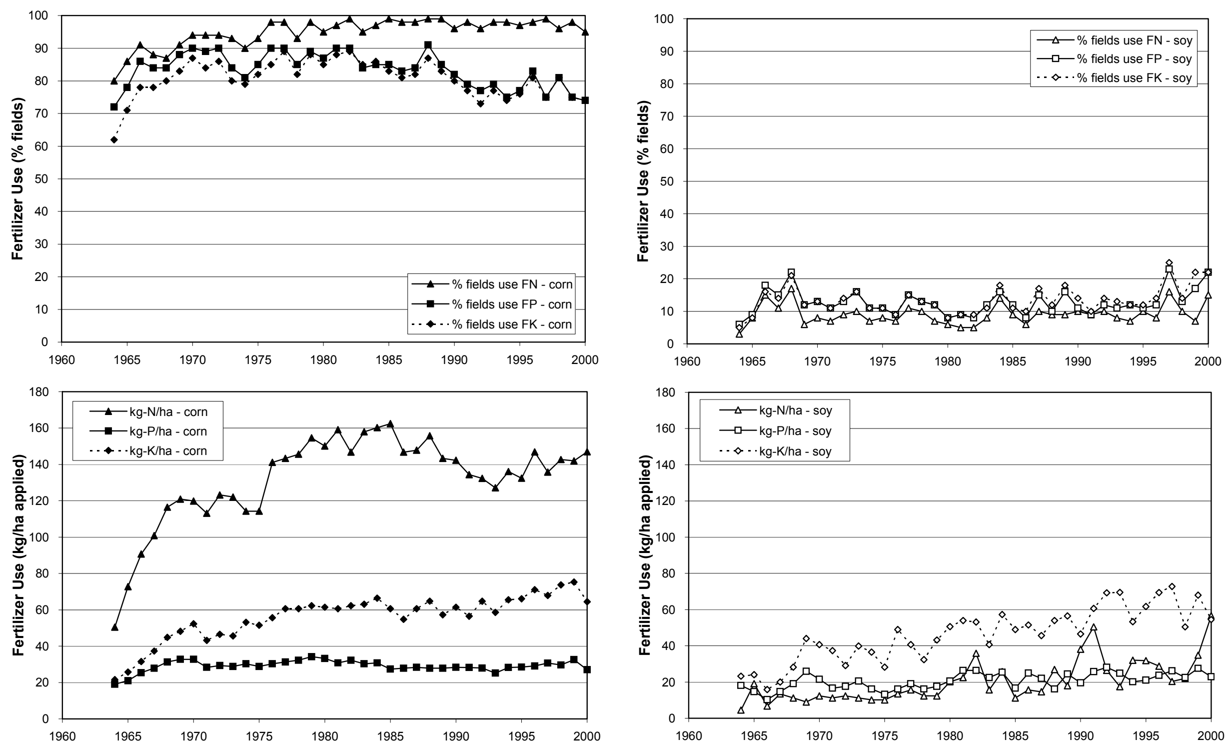


Figure 2. Historical rates of nitrogen, phosphorus and potassium fertilizer use on soy and corn fields in Iowa
(USDA Economic Research Service Fertilizer Use and Price Statistics Report (86012) (http://usda.mannlib.cornell.edu/) for 1964-1990 data and USDA NASS Quick Stats for 1990-2000 data (http://www.nass.usda.gov:81/ipedb/))

Table 1. Nitrogen Transformation Reactions in an Agricultural System

Type	Chemical Reaction[a]	Comments
Nitrification	$NH_4^+ + 1.5O_2 \rightarrow NO_2^- + 2H^+ + H_2O$ $NO_2^- + 0.5O_2 \rightarrow NO_3^-$	Nearly all NH_4^+ nitrified quickly. NO and N_2O generated as by-products
Denitrification	$5(CH_2O) + 4\ NO_3^- + 4H^+ \rightarrow 5CO_2 + 2\ N_2 + 7\ H_2O$ (really a series of several steps - $NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2$)	Requires organic matter and an anaerobic environment. NO and N_2O generated as by-products
Ammonification	$NH_2{\bullet}CO{\bullet}NH_2 + H_2O \rightarrow 2NH_3 + CO_2$	Urea used here as an example of N-containing plant matter. Combination of ammonification and nitrification often defined as "mineralization."
Fixation	N_2 + plant matter $\rightarrow$ organic-N	Process limited to a few plant and microbial species, including soy.

[a] after Canter (1997), Mitsch et al. (1999)

Nitrogen released from agrosystems as NO_x or NH_3 to the atmosphere returns quickly (hours of days) to the earth's surface, typically as HNO_3, NH_4NO_3, or $(NH_4)_2SO_4$ (Galloway et al., 2003). In addition to contributing to air quality problems, this reactive nitrogen can contribute to the total flux of NO_3^- and acid to surface water bodies, thus contributing to regional eutrophication and acidification problems.

The natural nitrogen cycle has been significantly altered by human activities, including substantial increases in the fixation of N_2 by industrial processes into ammonium products that are applied to soil to increase crop production rates. The industrial transformation of non-reactive to reactive forms of nitrogen by the Haber-Bosch process increased from 0 before 1910 to more than 100 Tg N/yr in 2000. 85% of this was used for fertilization (Galloway et al., 2003).

Nitrogen found in surface or groundwater supplies in the Midwest has likely come from commercial fertilizer application or manure associated with animal feed lots. Nitrogen is typically applied to row crops in the spring as anhydrous ammonium or ammonium nitrate. Past farming practices included fall fertilization when the fields were dry. Due to mineralization and leaching losses, however, this practice has largely been abandoned. The overall fate of the nitrogen in an agricultural setting is shown in Figure 3. Within a few weeks, most of the ammonia added to the soil is converted to nitrate, which can be taken up by plants, denitrified, or leached into water. The highest leaching rates occur in the spring after fertilizer is applied and rainfall is heaviest. Later in the summer, the growing crops effectively uptake much of the nitrate causing water and soil concentrations to decrease. Denitrification rates are also highest during these warmer months. Denitrification can reduce nitrate concentrations in both ground and surface waters. A USGS study in Eastern Iowa showed that 92% of the total nitrogen found in surface water is in the form of nitrate. Nitrate is considered to have a relatively short half-life in surface water bodies due to its transformation by denitrifying bacteria and uptake into plant matter where it then cascades through the food web. Galloway et al. (2003) estimate that 30-70% of nitrate that enters rivers and streams is eventually denitrified (1-20% in a particular reach of a river).

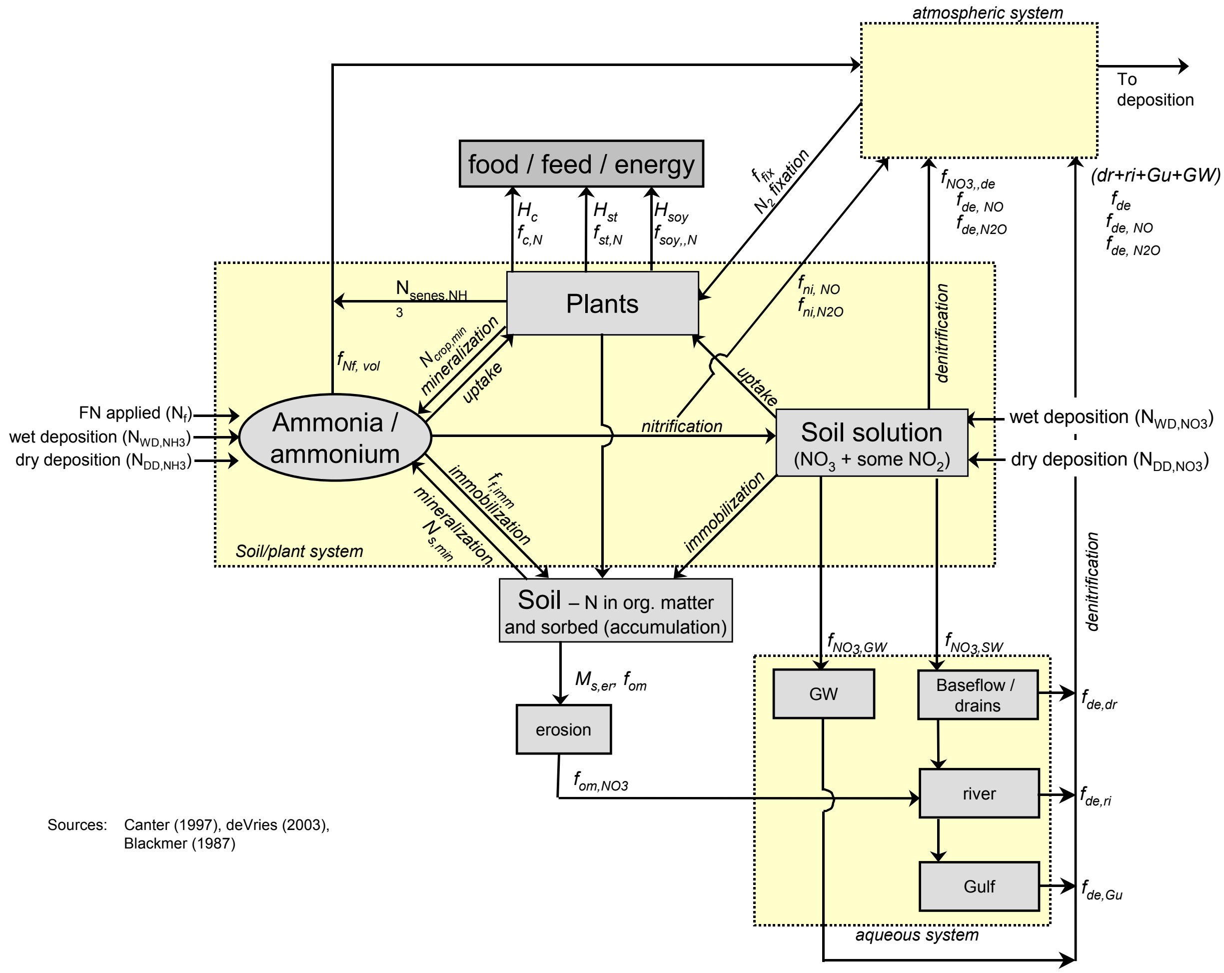


Figure 3. Major nitrogen transformation processes and flows in an agricultural system (N_i – Flows (kg/ha), f_i – fraction of flow)

Phosphorus Cycling and Fate in the Environment

Phosphate in the environment occurs most abundantly as calcium phosphate minerals, such as hydroxyapatite ($Ca_{10}(PO_4)_6(OH)_2$), which have a very low solubility (Smil 2000). Because phosphorus is often a limiting nutrient for plant growth, man-made inorganic phosphate fertilizers are now produced from these phosphate rocks and strong acids for application to agricultural lands (Smil 2000). A commonly used fertilizer, triple superphosphate ($Ca(H_2PO_4)_2$), contains 20% P. Global consumption of P fertilizers increased to over 15 Mt P yr^{-1} in 1990, but has decreased some in the last decade. This use of inorganic P fertilizer has contributed to Smil's (2000) estimate that the global mobilization of P tripled by the year 2000 relative to its natural flows.

Once phosphate fertilizer (FP) is applied to the soil, chemical equilibria with Al, Ca and Fe minerals in the soil dictates that much of FP reacts first to form insoluble mineral complexes that are very stable (Hansen et al., 2002). This sink must be satisfied to enable availability of excess P in the soluble and reactive pools necessary for plant uptake. Figure 4 identifies these forms of P and the important flows that distribute P among environmental compartments. Soil testing that extracts and measures only the bioavailable P (in soil solution and reactive pools) is required to determine optimal rates for adding FP to a field.

Phosphorus cycling and transformations in the environment are less complex than nitrogen cycling. There is only a minuscule amount of P in the atmosphere, so this environmental compartment can be left out of analysis of P cycling (Smil 2000). There are, however, also far less data available to quantify flows between these compartments, including weathering, dissolution and immobilization, in comparison with that available for nitrogen (Smil 2000).

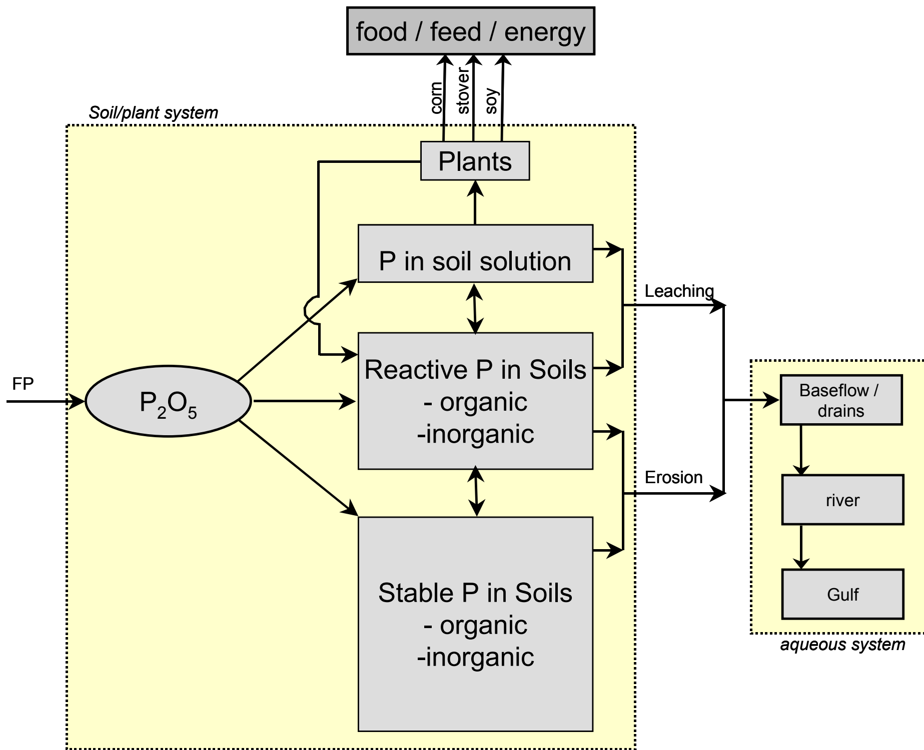


Figure 4. Major phosphorus transformation processes and flows in an agricultural system (adapted from Smil, 2000; Hansen et al., 2002; Sharpley et al., 2003)

An estimated 1-2% of P fertilizers dissolve into water (range of 0.1 – 5%) (Smil, 2000). Leaching of excess fertilizer into water results in transport of the soluble form – orthophosphate (PO_4^{-3}), primarily to surface waters (Becher et al., 2001). Most phosphorus sorbs to sediments and 70-95% of the total phosphorus (TP) load is transported through surface water bodies with suspended solids (Becher et al., 2001). Although this phosphorus is slowly desorbed into the water column, the bound form is generally not directly biologically available.

P is an essential mineral requirement for life and often a limiting nutrient for crop or algae growth. Corn has one of the highest intake rates, as much as 45 kg P ha^{-1} (Smil 2000). In soil microorganisms, the ratio N:P is on the order of 12:1 and for marine phytoplankton it is ~16:1 (Smil, 2000), based on an average composition of aquatic organisms ($C_{106}H_{263}O_{11}N_{16}P$) (Stumm and Morgan, 1990). In eutrophic waters, however, the N:P molar ratio in algae can be as high as 22:1 (Kalff, 2003). Thus, small additions of P can add sufficient nutrients for a large quantity of excess growth in an aqueous system.

Water Quality Impacts Associated with Nutrients

Hydrologic Conditions in Midwestern United States

The flow of water through the agricultural systems in the Midwestern United states plays a significant role in the transport of nutrient pollution to water bodies and its distribution between ground and surface water. The eastern half of Iowa receives an average of 800 to >900 mm rainfall each year. The highest rainfall rates are in the southeastern region of the state with less rainfall in the northwest (Brenner et al., 2001). Rain is sufficient in the eastern part of the state so that irrigation generally is not needed for row crop production.

The drainage of water from agricultural fields in the Midwest has been substantially altered by the installation of subsurface tiles and drains. The installation of these drains began in the early 1900's to make the region more suitable for crop growth. Overall, 25% of Iowa cropland is drained by subsurface tiles (USDA, 1987). Essentially, the tile drainage system results in most water entering streams and creeks results from ground water base flow and tile drainage rather than surface runoff.

There are two main reasons agricultural drainage systems are installed in the Midwestern and Great Lakes states: (1) to allow timely seedbed preparation, planting, harvesting, and other field operations; and (2) to protect field crops from extended periods of flooded soil conditions. Factors that make drainage a necessity for agricultural production include low soil permeability, flat or pot-holed topography, or periods of excess precipitation. (Zucker and Brown, 1998). Subsurface drainage can be considered a management tool to reduce the potential for erosion and phosphorus enrichment of surface waters. However, nitrate-N loadings exported from drainage conduits to surface waters continue to be a major water quality concern.

The extensively studied Walnut Creek watershed in central Iowa is a good example of the flat, pot-holed topography with extensive tile drainage. In the period 1992 – 1997, a total of 27% of the total precipitation drained through the subsurface tiles (Bakhsh et al., 2004). On an annual basis, between 12 and 50% of the total annual rainfall was transported to surface water through the tile drainage system. The higher percentages correspond to wetter years with corresponding very high nitrogen leaching rates.

In contrast to the low permeability of much of eastern Iowa's croplands, the major river systems in Iowa create valuable and high permeability alluvial aquifers. These sand flood plain aquifers are hydraulically connected to the rivers, leading to transport of contaminants back and forth between these systems depending on recharge versus discharge conditions. The floodplain aquifers are often prime agricultural land. The use of agrichemicals, rapid infiltration rates and shallow unconfined nature of alluvial aquifers makes them very susceptible to contamination by nutrients and herbicides (Canter, 1997).

Surface Water Contamination

Reference points to define potential impacts

Since both nitrogen and phosphorus are required nutrients in surface water bodies, some reference point is required to allow us to understand if anthropogenic inputs are truly a problem. Background concentrations, ambient water quality criteria, and total maximum daily loads (TMDLs) are all reasonable reference points to identify the severity or impact of anthropogenic nutrient loads. Table 2 identifies reference points that can be used to determine if the nutrient concentrations in Midwest ground or surface waters are unacceptably high.

Water quality and drinking water standards have been defined for nitrogen and phosphorus and can be used as a reference point to define the extent of impact from nutrients used in agriculture. The US EPA set a drinking water quality Maximum Contaminant Level (MCL) of 10 mg/L, primarily to protect infants from methemogloinemia (blue baby syndrome). The US EPA water quality standard for total phosphorus has been set at 0.1 mg/L to prevent excessive growth of aquatic plants in water bodies. Water quality criteria for nutrients were recommended by the US EPA in 2000 (2000a, 200b) for different eco-regions within the United States. These recommended criteria are presented in Table 2 for the Cornbelt and Northern Great Plains nutrient eco-region that encompasses much of Iowa. Although not directly comparable due to differences in units, the nitrogen water quality criteria for ecosystem health (TN) are less than the drinking water MCL (NO_3-N) that was set for human health. Since nitrate is the most common form of nitrogen in water bodies, the recommended water quality criterion for ecosystem health can be considered to be significantly more stringent than the MCL for human health. The recommendations are slightly less for TP than the current ambient water quality standard. Water quality criteria are consistent with concentrations typically used as the cut off for defining a river or stream as eutrophic (TN > 1.5 mg/L and TP > 0.075 mg/L (Porter, 2003)).

Recent efforts to identify background concentrations and loads to surface water bodies have been undertaken to support and justify recommended ambient water quality criteria. Nutrients are added to waterways in several non-anthropogenic ways – nutrient cycling by native plants and animals, increased loads after forest fires, and atmospheric deposition. Smith et al. (2003) used statistical analyses to estimate the natural background concentrations of nutrients in the 9 eco-regions identified for the nutrient water quality criteria (US EPA 2000a). Table 2 summarizes the estimated background concentrations and fluxes for the Corn Belt and N. Great Plains eco-region that encompasses much of Iowa.

Table 2. Reference Points to Identify Excessive Levels of Nutrients in Midwest Surface Waters

Potential reference point	Concentration (mg L^{-1})	
	TN	TP
Background levels	0.13 (0.105 - 0.19) [a,b]	0.021 (0.019 - 0.022) [a]
Recommended water quality criteria[c]	2.18	0.0763
Ambient water quality standard	--	0.100
Drinking water quality MCL	10 (as NO_3-N)	--
Definition of eutrophic[d]	1.5	0.075

[a] Frequency distribution (median (25% - 75%)) of background nutrient concentrations in the Corn Belt and N. Great Plains Eco-region (from Smith et al., 2003)
[b] corrected to remove atmospheric deposition
[c] Recommended water quality criteria, rivers and streams in Corn Belt Eco-region (US EPA 2000a, 200b)
[d] Porter (2003)

Extent of nutrient contamination in surface water

Surface water in the Midwest generally has concentrations of nutrients that are much higher than considered acceptable based on background concentrations or water quality criteria or standards. These concentrations are also highly correlated to agricultural practices and the high rates of fertilizer used on row crops.

The USGS sponsored a NAWQA (National Ambient Water Quality Assessment) study in Eastern Iowa during 1996-1998 (Becher et al., 2001). Samples were collected from both small watersheds (~300 km^2) to larger river watersheds, including the Cedar (20,200 km^2), Iowa (32,400 km^2), Wapsipinicon (6,050 km^2) and Skunk (11,200 km^2) Rivers. The area includes 64% row crop agricultural lands and 88% agriculture overall. Representative results are included in Table 3. In summary, the studies showed:

- NO_3-N concentrations:
  - ranged from <0.05 to 22 mg/L, with a median concentration 6.6 mg/L
  - 22% of the nitrate samples exceeded the MCL of 10 mg/L
  - samples collected directly from tile discharges were consistently > 10 mg/L
- Phosphorus concentrations:
  - ranged from 0.01 – 3.4 mg/L TP
  - 75% of the all of the samples exceeded the water quality limit of 0.1 mg/L for TP.

Both median N and P concentrations increased annually over the three-year study period in a manner that was highly correlated to increases in stream flow over the same period. Monthly variations showed the highest concentrations in May and June, presumably due to recent fertilizer application and high rainfall rates. These concentrations were often higher than the MCL.

Table 3. Nutrient contamination in surface waters in the Eastern Iowa NAWQA study

System description [a]	Drainage area (km^2)	% Agric. land	Median concentration (mg L^{-1})	
			TN	TP
NAWQA – Eastern Iowa basin 1996-1998	49,650	88.4%	7.2 (4.4 – 10.0) [b]	0.22 (0.1 – 0.34) [b]
Wapsipinicon River	6,050	87.4%	6.7	0.23
Iowa River	32,400	89.0%	6.0	0.30
Skunk River	11,200	87.1%	6.8	0.33
Flood Creek	320	95.3%	8.5	0.095
South Fork Iowa	580	95.1%	10.0	0.072
Wolf Creek	770	95.6%	10.5	0.13

[a] Becher et al. (2001) examples among 12 watersheds analyzed
[b] (25^{th} – 75^{th} percentile)

A longer-term study of nitrate concentrations in the Raccoon River in central Iowa provides a more clear understanding of the variability of nitrate concentrations and loads as a function of season and rainfall (Lucey and Goolsbey, 1993). Nitrate concentrations generally varied on a seasonal cycle with nitrate loads (typically 0 - ~300 mt/d with high of 780 mt/d) highly correlated to specific rainfall events, as characterized by daily variation in the streamflow.

In the Iowa River, NO_3-N concentrations have tripled since 1907, when they were <1 mg/L (Libra, 2001). Both the land area dedicated to row crops and the rate of nitrogen fertilizer used have increased significantly over that time. Others have also recognized that, over a wide range of scales, the mean NO_3-N concentrations (mg/L) (as calculated over several years in the 1990's) are proportional to the percentage of total area that is planted in row crops (RC) in both small and large watersheds (Goolsby et al., 1999; Schilling and Libra, 2000; Becher et al., 2001). For example, for small watersheds (47 - 775 km^2) in Iowa, Schilling and Libra (2000) determined that the nitrate concentration could be estimated as:

$$NO_3\text{-}N = 0.111\ RC + 0.217 \quad (R^2=0.94) \qquad [1]$$

The slope of the line, which is generally around 11%, decreases with increasing scale of the watershed, presumably due to dilution and denitrification processes that occur within the stream (Schilling and Libra, 2000).

Due to the low solubility of phosphorus in water, phosphorus concentrations for the NAQWA study area were observed to be most closely associated with the concentration of suspended solids in the sample (Becher et al., 2001):

$$TP = 0.0005\ C_{sed} + 0.2317 \quad (R^2=0.62) \qquad [2]$$

where the TP and sediment (C_{sed}) concentration are in mg/L. The suspended solids concentration is in turn related to agricultural management practices and the intensity of rainfall events.

Hypoxia

High nutrient concentrations contribute to eutrophic rivers and lakes at a local scale as well as at the National scale. Hypoxia is the condition defined by low oxygen levels (< 2 mg/L) that is caused by excess biological growth and decay. This condition often occurs in deeper water as the result of a coupling of the algae decay and seasonal water stratification due to temperature gradients. The region affected by hypoxia off the Louisiana coastline has increased from less than 10,000 km^2 in the mid-1980s to an average of 18,000 km^2 in the mid-1990s (Goolsby and Battaglin, 2000).

The hypoxia zone in the Gulf of Mexico has been recognized as a significant environmental impact. In general, biological productivity and fishery yields drop dramatically after a water body enters into a hypoxic condition. One study completed by the 1999 Integrated Assessment of Hypoxia in the Gulf of Mexico task force assessed the environmental and economic consequences of hypoxia (Diaz and Solow, 1999). Although there was not a lot of research completed to support this study, they observed hypoxia related stress in this region, including benthos mortality, elimination of larger longer lived species (who are suspected to have fled the area), and a shift in biological activity to non-hypoxic seasons. The economic assessment, however, did not find any direct effects attributable to hypoxia, although it is acknowledged that these effects might be masked by the general variability in fishery yields (Diaz and Solow, 1999).

Nutrient loads to the Gulf of Mexico are largely attributed to high nitrogen fluxes from agricultural practices in the Midwestern United States, coupled with increased total water discharges (+30%) from the Mississippi River and decreased residence time due to channel straightening and floodplain/wetland losses. The changes in the hydrologic system decrease the opportunity for denitrification and uptake of N in organic matter that accumulates in the sediments (Galloway et al., 2003), thus contributing to the overall increased transport of nitrate to the Gulf of Mexico. Eventually, most of the nitrate transported through rivers, lakes and coastal waters is denitrified (Galloway, 1998).

Howarth (1998) estimates increases in the nitrate loads of 2.2 – 6.5 times the estimate for pristine conditions. Goolsby and Battaglin (2000) estimate a similar increase; nitrate-nitrogen loads discharged with the Mississippi River have tripled from ~0.3×10^6 metric tons in the late 1950s to an average of approximately 1×10^6 metric tons per year since 1980. An additional 0.58×10^6 metric tons are discharged in the form of organic nitrogen, for a total average annual load of 1.6×10^6 metric tons (Goolsby et al., 1999). These increases in the reactive nitrogen load are correlated to increases in crop production and associated chemical fertilization rates in the heartland of the US. Fertilizer is clearly the most significant anthropogenic source of nitrogen to the Mississippi River (Table 4). Note that the sum of the total fluxes to the Mississippi River greatly exceeds the total load to the Gulf of Mexico due to denitrification and other nitrogen sinks that decrease concentrations as water flows from the source towards the Mississippi and the Gulf.

Table 4. Sources of Nitrogen to the Mississippi River

Source	**Nitrogen Input** [a] (millions metric tons)
Soil mineralization	6.8
Fertilizer	6.8
Legumes and pasture	4.4
Animal manure	2.8
Atmospheric deposition of nitrate	1.1
Atmospheric ammonia	0.7
Municipal and industrial point sources	0.3

[a] (averages for 1990-1996; Goolsby and Battaglin 2000)

Phosphorus loads to the Gulf of Mexico are substantially less than nitrogen loads. The average annual discharge from 1980 to 1996 was 136,000 metric tons. Most of this (69%) is transported as part of the sediment load, the remainder is primarily as dissolved orthophosphate (Goolsby et al., 1999). These loads have not changed substantially since 1970.

Row crop production in the Midwest contributes the highest fluxes of nitrogen and phosphorus to the Mississippi River basin. As shown in by the area denoted as region 6 in Figure 5, Eastern Iowa, Illinois and the surrounding vicinity, have the highest average yields (mass per land area per year) within the

Mississippi drainage basin of nitrate (650-1150 kg km^{-2} yr^{-1}), total nitrogen (1000-1690 kg km^{-2} yr^{-1}), total phosphorus (96-129 kg km^{-2} yr^{-1}), and orthophosphate (35-36 kg km^{-2} yr^{-1}). These yields represent average calculated yields over the period 1980 to 1996. These years include both very wet (1993) and very dry (1988) years.

Variability in the nutrient loads to the Gulf of Mexico is highly dependent on the annual mean river discharge (Figure 6), which in turn is a direct consequence of the variability in the annual rainfall in the Midwest each year. The higher nutrient loads observed during wet years are also somewhat correlated to the size of the hypoxic zone in the Gulf of Mexico (Figure 7). The trends are very apparent for 1985 – 1993. The hypoxic zone remained large, however, after the heavy flooding in 1993.

A model developed by Scavia et al. (2003) suggests that the size of the hypoxic zone in the Gulf of Mexico is correlated to the TN load to the Gulf during May – June. The size is estimated based on this TN load and simplified ocean mixing dynamics. The model predicts that a 40-45% reduction in the average TN load (1980-1996) would be required to maintain the hypoxic zone at <5000 km^2 – a goal specified by the NOAA Integrated Assessment of Hypoxia in the Gulf of Mexico (Brezonik et al., 1999). This required percentage reduction is greater than the 20-30% reduction in TN load suggested by the task force itself (Brezonik et al., 1999).

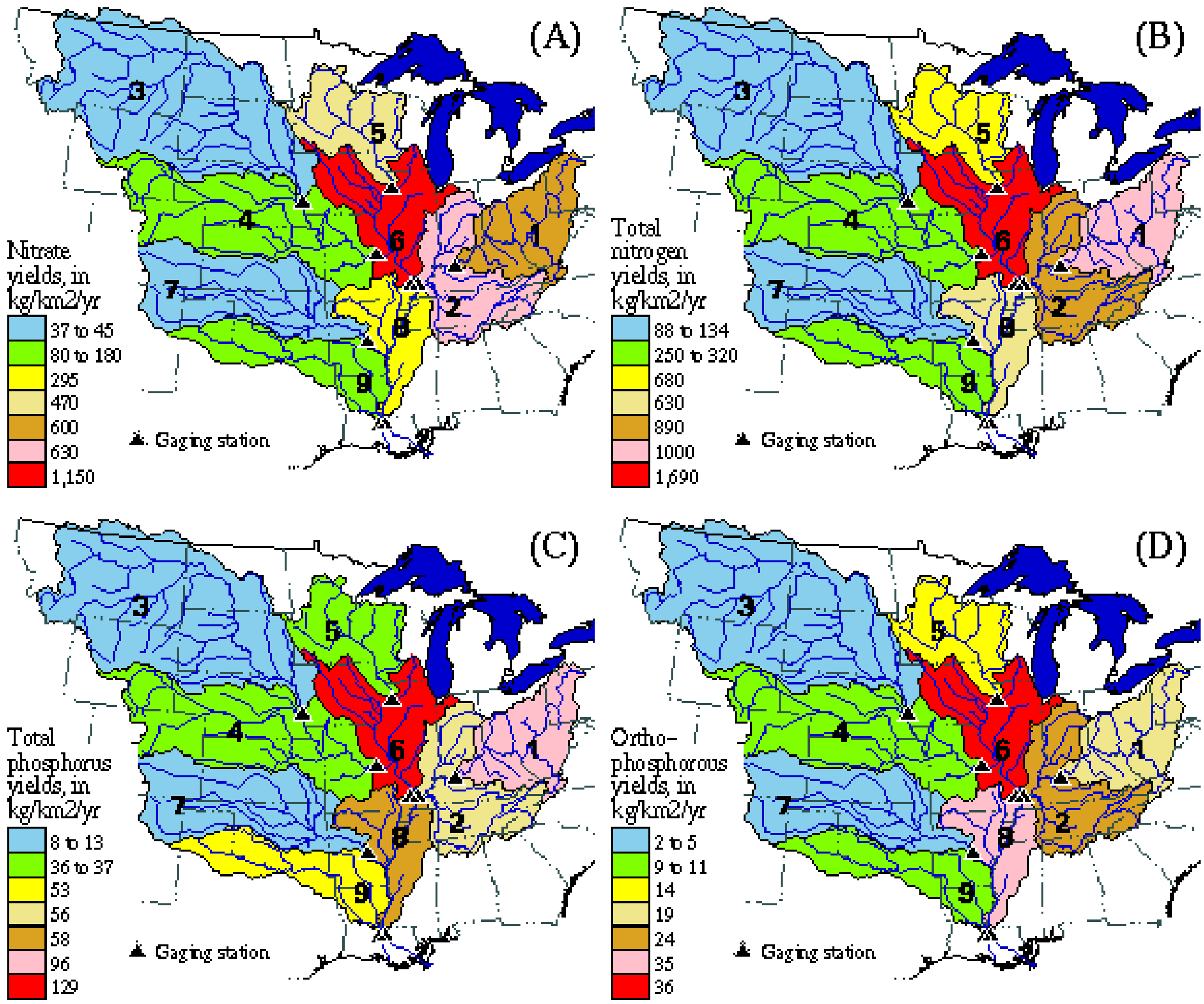
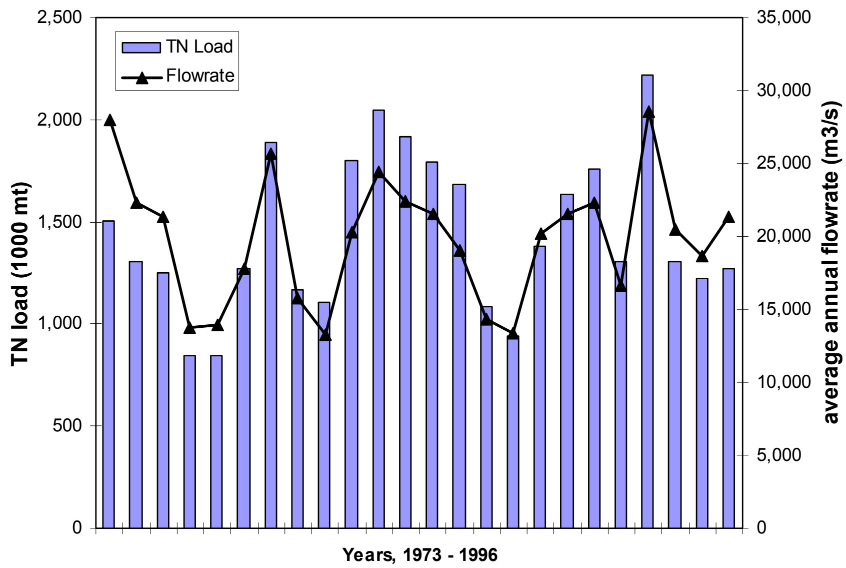
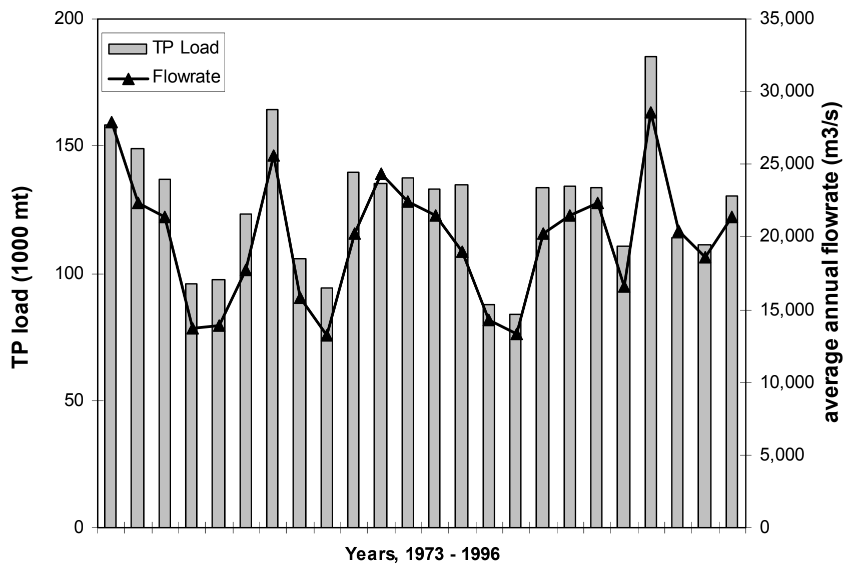


Figure 5. **Yields of nitrogen and phosphorus contributing to Hypoxia in the Gulf of Mexico** (Goolsby et al., 1999)



a)



b)

Figure 6. Total discharge of (a) nitrogen and (b) phosphorus from the Mississippi River to the Gulf of Mexico are highly correlated to the total river discharge
(data from Goolsby et al., 1999, NOAA report http://co.water.usgs.gov/hypoxia/html/nutrients_cenr.html))

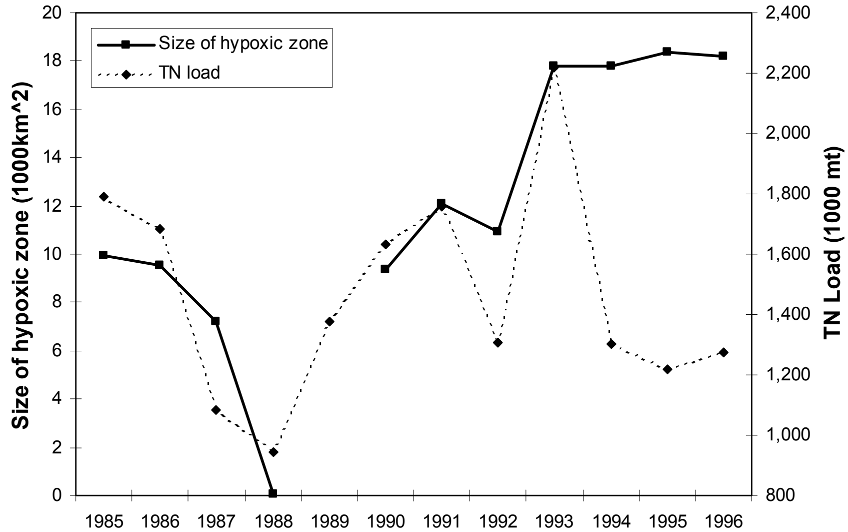


Figure 7. The size of the hypoxic zone varies substantially among years and is correlated to the TN load from the Mississippi River.
Note – no data are available for 1989. (data from Goolsby and Battaglin, 2000 (USGS Fact Sheet 135-00); NOAA report, http://co.water.usgs.gov/hypoxia/html/nutrients_cenr.html)

Ground Water Contamination

The impacts of groundwater contamination in the Midwest include contaminated drinking water supplies and subsequent surface water contamination as shallow groundwater discharges to surface water bodies. Thus, there are several different reference points that could be used to define this extent of contamination.

- Natural background nitrate levels in groundwater for Iowa <2 mg/L NO_3-N (Kross et al., 1993)
- Drinking water quality MCL (10 mg/L NO_3-N)
- Recommended ambient water quality criteria (Table 2)

Based on any of these reference points, the ground water in Iowa and much of the Midwest is considered highly contaminated. A statewide analysis of the quality of drinking water in private drinking water wells was sponsored by the Iowa Dept. Natural Resources in the late 1980s. Kross et al. (1993) report many wells exceed the MCL of 10 mg/L NO_3-N. These include:

- 18.3% of all private wells;
- 35% of all shallow wells (<15 m) (with 15.6% of shallows well having NO_3-N >20 mg/L); and,
- 9.2 – 11.6% of wells in eastern Iowa and 38.2% of wells in NW Iowa.

Nitrate, the nitrogen form that is most abundant in ground water systems, is now considered nearly ubiquitous in shallow Midwestern groundwater at concentrations considered to be anthropogenic (>3 mg/L). It was detected 13 times as often as NH_4^+ or organic nitrogen in ground water samples (Nolan and Stoner, 2000). Drinking water supplies from deeper ground water are generally much safer due to the increased age of ground water. In eastern Iowa, with wells for public drinking water supplies at 130-190 ft deep, median NO_3-N concentrations are low (<0.1 – 1.3 mg/L). Long residence times, which allow for denitrification and/or the possibility of ground water age pre-dating significant commercial fertilizer applications, contribute to these lower concentrations (Nolan et al., 1997).

Burkart et al. (1999) used multivariate analyses and data from over 300 wells in the Midwest to identify site characteristics that are most closely correlated with nitrate concentrations. Low nitrate concentrations were most significantly correlated to a seasonally high water table. This condition is generally indicative of poorly drained soils that cause an increase percent of water to flow to surface water bodies and anaerobic conditions that increase denitrification rates.

Results from the USGS National Ambient Water Quality Assessment (NAWQA) program showed that areas with high nitrogen yields (>21 kg/ha), well drained soils, and low woodland to cropland ratios (<0.3) are the most susceptible to nitrate pollution in shallow groundwater (Nolan et al., 1997). Areas with all three of these factors are defined as the highest risk group. This group had median concentrations of 4.8 mg/L NO_3-N, with 25% of samples exceeding the MCL. In Iowa, the entire state was considered at risk due to high nitrogen inputs. With differences in the nature of the infiltration rates, however, approximately 50% of the state is in the highest risk category and 50% in the next highest. Poorly drained soils and the agricultural practice of constructing subsurface tiles greatly reduce the potential for the applied nitrogen to enter ground water systems in much of Iowa.

Burkart and Stoner (2002) add irrigation to the list of factors that contribute to high risk. Irrigated lands are often well drained and the combination of a large amount of transport medium (irrigation water) and rapid transportation rates results in a significant fraction of the applied nitrogen reaching groundwater resources. Nebraska and other row crop agricultural areas have a much higher risk of ground water contamination by nitrates than eastern Iowa.

The contamination of groundwater systems by nitrate represents a long-term loss of this water resource. Nitrate concentrations are slow to respond to measures instituted to prevent further contamination, and may take decades to recover. Tomer and Burkart (2003) measured NO_3-N concentrations in two small watersheds in SW Iowa. The watersheds were nearly identical geologically and both had continuous corn grown from 1964-1974. One watershed, however, had very high nitrogen application rates for 1969-1974 (446 kg ha^{-1} yr^{-1}). In 1996, the crop rotations were changed and a study instituted to determine the impacts of this change on groundwater quality. Through analysis of the age of ground water sampled, they show, however, that the ground water concentrations are still greatly influenced by the high nitrogen application rates in the early 70s. Decades might be required to experimentally verify and changes in agricultural practices on groundwater quality due to the long-term impacts of past practices on ground water quality.

Variables Affecting Nutrient Leaching to Surface Water

Fertilizer application rates, tillage practices, crop rotations and rainfall have all been considered as potential causes of variability in the amount of nutrients that leach as non-point source pollution from agricultural fields to surface water bodies. Since the solubility of nitrates and phosphates are quite different, it is possible that these other variables will affect the loss of these nutrients to surface water differently.

Nitrogen

Early efforts to identify the most important variables for nitrate runoff suggested that the amount leached was directly proportional to the fertilizer application rate (e.g., Hallberg 1987). These results were based on short-term experiments (2-3 years) with very wide ranges of fertilizer application rates (0 – >400 kg-N/ha) (e.g., Gast et al., 1978). Over a more realistic fertilizer application rate, at least for corn, of 100-200 kg/ha, other variables have been found to be more important than the fertilizer application rate.

Studies of variability in nitrate losses through subsurface drains indicate that differences in rainfall and the amount of water transported through subsurface drains are significantly more important variables than differences due to cropping or tillage practices. Figure 8 illustrates the results from Weed and Kanwar (1996), who show there is no consistent trend or statistically significant differences in the fraction of FN that leached into drainage tiles between the C-S rotation with conventional till and either continuous corn (C-C) or C-S with a no till (NT) operation. At this site near Nashua IA, much more substantial differences occur among the years due to differences in rainfall than among the crop rotations or tillage practices.

Later studies at the same site also show that differences in the amount of nitrate leached among the years (5-59% of FN applied for C-S, chisel plow (CP) system) due to rainfall are more significant than due to fertilizer application or tillage practices (Bakhsh et al., 2002). They observed no statistically significant differences between corn and soy rotations for nitrate leached (kg/ha). A significantly higher average fraction of nitrate was leached from chisel plow fields (21%) than no till (NT) fields (14%) when nitrogen was applied in the spring at a rate based on soil test results. For fall application of FN at the same rate on all fields, however, the average fraction of FN leached from the no till plots (41%) was significantly higher than for CP fields (24.5%).

With the most significant overall variability in nitrate leaching rates among the years due to differences in rainfall, Bakhsh et al. (2002) found that they could describe 79% of the total variability based on linear relationships that show the total nitrate leaching loss as a direct function of the subsurface drainage. Drainage, in turn was linearly related to rainfall. The net relationship shows:

$$NO_3\text{-N loss (kg/ha)} = 0.070 * \text{rain (mm)} - 41.72 \qquad [3]$$

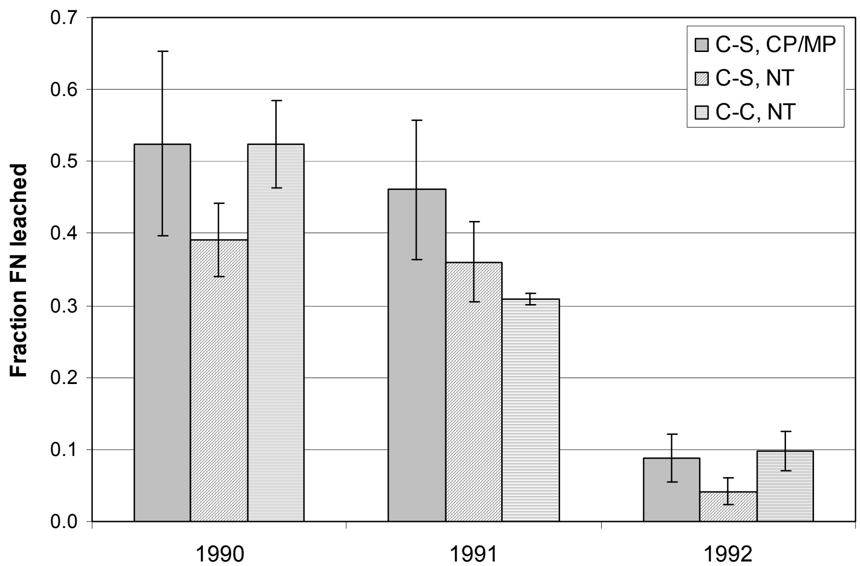


Figure 8. During a carefully controlled field study, Weed and Kanwar (1996) found that there were no statistically significant differences in the fraction of nitrate leached from C-S and C-C rotations under different tilling operations. (MP – moldboard plow)

Phosphorus

Data quantifying TP losses at field and small watershed scales are scarce. One study by Klatt et al. (2003) measured P inflows to Clear Lake in NE Iowa over a two-year period. Clear Lake is within the geographic boundary considered here and is a good example of a water body adversely impacted by eutrophication. The TP load to the lake and TP used within the watershed are presented by Klatt et al. (2003). As was observed with nitrogen loads to surface water, the TP loads were significantly higher during the year with a higher total rainfall.

Statistically significant differences have been observed for phosphorus loads as a function of tillage practices, with higher loads often observed from no till plots. Gaynor and Findlay (1995) observed an factor of two increase in the TP loads from fields with no till practices compared to conventional till.

McIsaac et al. (1995) also observed this increase for soluble phosphorus loads. They observed increases in SP to be 1.6 to 5.4 times higher for no till plots. The range of values was a function of different soil types. Daverede et al. (2003) observed a 3-fold increase in dissolved reactive phosphorus loads for NT, but no statistically significant difference for TP loads.

These increases in phosphorus loads were unexpected. In general, NT reduces erosion substantially compared with CP practices. And, since most P is bound to eroded soils, it was thought that NT would reduce TP loads. The increased availability of P from crop residue decaying at the ground surface and the formation of root channels and other macro pores in the subsurface provide avenues for increased transport of soluble phosphate that is more substantial than the beneficial effects of reduced erosion.

In general, phosphorus losses at the field scale are estimated in an empirical fashion based on a combination of P source factors (reactive P concentration in the soil, FP application rate and method) and transport potential (erosion, runoff, percolation) (Birr and Mulla, 2001; McDowell and Sharpley, 2001; Sharpley et al., 2001). The Mehlich-3 extraction process is typically used to define reactive P concentrations. This empirical approach provides a relative index to define vulnerability; it is not intended to predict phosphorus loads to surface or ground water bodies.

Modeling Fertilizer Nutrient Discharges to Surface Water

A wide variety of approaches has been used to estimate non-point source nutrient discharges from agricultural fields to surface water bodies. These approaches can be broadly classified as:

- Empirical emission factors that base nutrient release on a fraction of the nutrient input, with or without stochastic distributions of the emission factors and input parameters
- Mass balance approaches that assume nitrogen, which is not already incorporated into plant matter or soils, or the atmosphere, has the potential to leach
- Mechanistic modeling of nutrient flows and reactions within and among environmental compartments.

Others have also used hybrid approaches that combine aspects of the various approaches. Clearly, these approaches encompass a diversity of model complexity, suitable scale of application, and input data requirements. The empirical emission factor approach is the easiest to apply, and could be appropriate for simple systems such as phosphate discharges to surface water. However, it is least likely to capture the complex interactions among various transformations and partitioning of nitrogen among environmental compartments, especially as a function of the variety of agricultural practices and geologic, hydrologic and weather conditions among different farming regions. At the other extreme, mechanistic modeling efforts require significant amounts of site specific data, but provide more accurate estimates of nutrient fate and flows and their variability. The integration of GIS (geographic information systems) with these mechanistic models provides improved data management capabilities, allowing mechanistic models to be applied over larger scale systems.

Emission Factor Approaches

The model developed by Wang (1999) to estimate greenhouse gas and other air emissions associated with fuel use for transportation is a good example of a model based on emission factors. This model, GREET, estimates most nitrogen flows associated with feedstock preparation of biomass fuels as a fraction of the fertilizer nitrogen used. This includes atmospheric emissions of NH_3, NO_x and N_2O. Fertilizer leaching is estimated as 24% of the FN used. This fraction was determined from literature defining nitrogen leaching rates measured at several different field test plots. Summaries of these data (e.g., Gast et al., 1978; Hallberg, 1987), support the use of an emission factor that describes nitrogen leaching based on the rate of FN application. The GREET emission factor for FN was employed only to allow subsequent estimation of N_2O emissions as the leached nitrate is denitrified. The use of a distribution of values for the fraction of FN ultimately emitted as N_2O provides the only means of integrating variability in the leaching rates into the GREET model (triangular distribution from 1 to 2% with maximum 1.5% of FN transformed into N_2O).

The nitrogen flow modeling by the International Panel on Climate Control (IPCC, 1996) also focused on air emissions, and recognized the need to quantify leaching to surface waters due to the subsequent denitrification and release of N_2O. The IPCC suggests that nitrogen leached to surface waters could be estimated as 30% of the applied fertilizer nitrogen that remains after volatilization, although they recognize values could range from 10-80%.

Mass Balance Approaches

Many of the original mass balance approaches generally tried to estimate potentially leachable nitrogen based on the difference between quantifiable nitrogen inputs, outputs and accumulation of nitrogen in the soil system at a field scale (Meisinger and Randall, 1991; Schepers and Moiser, 1991; Follett, 1995; Shaffer et al., 1995). Inputs include fertilizer (inorganic and manure), atmospheric deposition, irrigation, soil and crop mineralization, and nitrogen fixation by legumes. Quantifiable outputs include nitrogen removed with the harvested crop, volatilization, erosion, plant senescencing (direct loss of NH_3 from plant tissue). Application of the mass balance approach to determine leaching by difference requires an assumption that the soil nitrogen concentration is at a steady state or direct measurement of soil-N concentrations to quantify the accumulation or depletion of soil nitrogen.

The mass balance approach can identify potentially leachable nitrogen, but does not adequately identify the distribution of the leached nitrogen between surface and ground water. Systems for which all of the water flow can be assumed to go to either surface or ground water are required for a mass balance to be used. For example, Puckett et al. (1999) was able to assume that all of the leached nitrogen was transported to a highly permeable outwash aquifer. The leaching rates were determined assuming a steady state between inflows and outflows and that no nitrogen was lost by runoff or base flow.

Mass balance approaches to identify nitrogen flows have also been applied to large watershed scale systems around the world. In many of these cases, the quantity of nitrogen and/or phosphorus leached was based on integration over time of measured river concentrations and flow rates. Several such studies have attempted to define empirical models for the river flux of N or P based on some measure of nutrient inputs (Howarth et al., 1996; Carpenter et al., 1998; Caraco and Cole, 1999; Goolsby et al., 1999; David and Gentry, 2000; Carey et al., 2001; McIsaac et al., 2001; McIsaac et al., 2002). The anthropogenic nitrogen inputs (ANI) or net anthropogenic inputs (NANI) are often used as the independent variable in these models. The models range from globally applicable (Caraco and Cole, 1999), to those developed for a specific large watershed. McIsaac et al (2001, 2002) for example, developed a regression equation for the Mississippi River Basin. Their regression recognizes the importance of historical nitrogen inputs as well annual variation in rainfall:

$$N_{LM} = 0.66 \cdot W^{0.93} \cdot \exp(0.13 \cdot NANI_{2\text{-}5} + 0.06 \cdot NANI_{6\text{-}9}) \qquad [4]$$

where, W is the annual water yield (stream flow / watershed area) and NANI, the net anthropogenic nitrogen input (FN + atm. dep. + fixation – N in crops and feed), is quantified for the previous 2-5 years and 6-9 years. The model accounts for 95% of the variation in annual nitrate flux (N_{LM}, kg ha^{-1} yr^{-1}) at the monitoring station in the lower Mississippi River just before discharge to the Gulf of Mexico.

Similarly, Goolsby and Battaglin (2001) estimated nitrate fluxes to the Gulf of Mexico as:

$$N_{LM} = 0.049 \cdot L_{f,Y2} + 36 \cdot Q - 0.094 \cdot R_{Y1} \qquad [5]$$

where $L_{f,Y2}$ is the nitrogen fertilizer load (mt) 2 years previous, Q is the mean annual stream flowrate (m^3s^{-1}), and R_{Y1} is the residual nitrogen (inputs minus outputs) the previous year.

The highly non-linear nature of these equations makes it difficult to allocate the total nitrogen flux to specific sources.

On a much smaller scale, Tomer et al. (2003b) used similar types of regression analysis to estimate nitrate fluxes from tile drainage systems in Walnut Creek, central Iowa from 1992 through 2000. They found a log linear relationship between stream flow (Q, mm hr^{-1}) and the flux (kg ha^{-1} yr^{-1}).

$$N_{LM} = a \cdot Q^b \qquad [6]$$

where a ~ 4 and b ~ 1.1. Regression coefficients were statistically similar for the two tile systems, but different for the downstream sampling point. For each individual system, the R^2 values were very high (>0.99). Application of this equation at a larger scale would require flow data and spatial integration of this non-linear relationship over all tiles drainage systems.

The work of deVries et al. (2003) provides a hybrid approach based on both mass balance concepts and emission factors to quantify both aqueous and air nitrogen emissions. They essentially describe nitrogen cycling from agricultural and non-agricultural lands as a series of steps. Losses at each step then reduce the mass available for loss at the next step. The processes – in the order of assumed occurrence – include: ammonia volatilization, crop uptake, immobilization/mineralization, nitrification/denitrification, leaching to ground water, leaching to surface waters and subsequent denitrification, outflow to the sea. If at any point, the losses from previous steps exceed the nitrogen inputs, then there are no additional losses for the last processes in this series of steps. The flows and fractions lost at each of these steps were estimated for application in the Netherlands.

Mechanistic Models

Mechanistic models for nutrient flows attempt to explicitly model flows and interrelationships between these flows and nutrient use, crop productivity, weather patterns, and geographic conditions. Thus, the most comprehensive models are composed of submodels defining land use, soil quality, crop growth, hydrology, erosion and nutrient transformation and transport. Application of this type of models requires a vast set of site-specific parameters to quantify all of these interrelationships.

SWAT (soil water assessment tool) developed by the USDA Agricultural Research Service (Arnold et al., 1998) is one of the most comprehensive watershed models currently available, especially when linked to a model such as EPIC (erosion productivity impact calculator) that estimates field-scale crop growth, N and P cycling, runoff and erosion (Williams et al., 1989). SWAT, which models water, sediment and chemical flows in large watersheds, has been applied to watersheds in Story (Walnut Creek) and Poweshiek (Buck Creek) Counties in central Iowa to predict the effectiveness of changes in agricultural practices on the reduction in erosion and nitrate loads to surface waters (Vache et al., 2002). It is also a component of the computation tool currently recommended by the US EPA to help establish total maximum daily loads (TMDLs) to maintain adequate water quality (USEPA, 2001; DiLuzio et al., 2002), and has been used in this capacity for a growing number of regions (e.g., Santhi et al., 2001).

Wauchope et al. (2003) present a review of the numerous other mechanistic models could also be used to estimate the flow of nutrients or herbicides from agricultural fields. Some other examples include:

- GLEAMS – Groundwater Loading Effects of Agricultural Management Systems. This model includes hydrology, erosion, pesticide and nutrient transport for a homogeneous field scale system. It was originally developed to estimate leaching of pesticides and nutrients within and below the root zone. It has recently been used to estimate nutrient and herbicide losses to tile drains at a field scale in Iowa (Bakhsh and Kanwar, 2001). Although this model is no longer supported by the USDA, its equations describing the movement of pesticides through the subsurface were integrated into SWAT (Wauchope et al., 2003)

- RZWQM – Root Zone Water Quality Model. This model provides a one-dimensional dynamic tracking of nutrient and pesticide leaching to ground and surface water. This model has been successfully applied to a field in the Walnut Creek watershed to predict nitrate losses to subsurface drains (Bakhsh et al., 2004)
- AnnAGNPS Annualized Agricultural Non-Point Source model. This model extends AGNPS, which was developed to estimate non-point source discharges at a watershed scale based on the hydrology and erosion with individual storm events (Young et al., 1989). The new annualized version can use actual annual weather data or such data can be generated with an internal stochastic climate module. It is characterized by the developers as requiring less data than models such as SWAT. (USDA, 2004a)
- NAPRA WWW – National Pesticide Risk Analysis model for the WWW. Lim and Engel (2003) extended NAPRA to include nutrient leaching (from GLEAMS) and developed a WWW interface that provides relatively easy access to the required soils and hydrology data for any county in the United States. Output from this model includes a distribution of probabilities that water quality standards will be exceeded for a set of input agricultural practices.

Mechanistic models such as those described above have not been used in agricultural life cycle assessments to date. This is most likely due to the extensive parameterization required (soil types, weather, agricultural practices etc.), even for field scale systems.

3. Methods for Quantifying Lifecycle Inventory Data for Nutrient Flows Associated with Corn-Soybean Production

General Approach

The research reported here quantifies lifecycle inventory flows for corn and corn-soybean rotations, with and without stover removal. The focus of this work is on nutrients, although additional flows were also quantified to complete a cradle to farm-gate LCI for these systems. Inventory parameters required to assess environmental impacts associated with eutrophication, acidification, and greenhouse gas emissions were calculated to enable a more complete understanding of the relative environmental costs and benefits associated with the preparation of feedstocks for bio-based fuels and products.

The mathematical models describing the flows and distribution of nutrients among the various environmental compartments uses a hybrid mass balance and emission factor approach. That is, these flows are based on some fraction of a suitable nutrient inflow. Two differences, however, advance these techniques for application in an LCI:

- To the extent possible, the fractions integrate knowledge of the physical, chemical and biological mechanisms that control the flow. For example, NO emissions are based on a fraction of the nitrogen that undergoes nitrification or denitrification reactions, rather than a fraction of the total fertilizer nitrogen applied.
- Variability in the distribution of nitrogen among environmental compartments, especially leaching to surface waters, as a function of rainfall rates is included.

Data for many of the agricultural flows were taken from a specific geographic region that includes three major watersheds in eastern Iowa. The choice of this region allowed the calibration of nutrient leaching rates against water quality data available from the USGS. Temporal variation in rainfall is integrated through the inclusion of several years of data.

System Boundaries

Applying an agricultural LCA to a specific geographic location provides the increased accuracy by using site-specific data. A balance must be made, however between a site that is so small that it is not representative of the broader system versus a region that is so large that input data and nutrient flows vary substantially over the region and are difficult to characterize. The eastern Iowa region selected for this study encompasses approximately 50 thousand km^2. It was chosen based on its high productivity of corn and soybeans, the availability of agricultural practice, yield and erosion data, and its alignment with three major watersheds for which several years of water quality data exist. These data were essential for the development and calibration of nutrient leaching models.

The system also included thirteen individual years – 1988 – 2000. Both very wet (1993) and very dry (1988) years were observed in this period. Data for the 1990s was readily available electronically through the USDA National Agricultural Statistics Service (USDA NASS) databases. Depending on the nature of the data, information for earlier years was available through NASS reports or databases. The inclusion of a thirteen-year period allowed variability of crop yields and nutrient flows as a function of rainfall to be incorporated. The use of average data would have lost the important environmental impacts associated with extreme climatic conditions. Employing average values with statistical distributions in the values used could have integrated some of this variability, but it would be harder to capture the correlations between nutrient fate and rainfall.

The eastern Iowa region is shown in Figure 9. County boundaries were used to define the geographic system boundary to approximate the overall area encompassing the Wapsipinicon, Skunk, Iowa and Cedar River watersheds (Table 5). The Cedar/Iowa river systems provide the greatest contributions in terms of drainage area and the stream flow in this Eastern Iowa watershed region. There is over one order of magnitude variation in stream flow among years considered. For all the rivers, the lowest flows were recorded in 1989 and the highest in 1993. These trends correspond to annual rainfall extremes (Figure 10). This entire region is considered to be humid, especially in the southeastern sections where there is significantly more rain and warmer temperatures compared with the northern regions of Iowa.

Table 5. Characteristics of Eastern Iowa Watersheds

Watershed System	Drainage area (km^2)[a]	Annual average flow rate (m^3/s)[b] (average and range 1988-2000)	% drainage basin used for agriculture[a]	% drainage basin used for row crops[a]
Wapsipinicon R.	6,050	57.6 (10.7 – 149)	87.4	70.3
Iowa R. (including Cedar)	32,400	300 (51.8 – 850)	89.0	65.3
Skunk R.	11,200	88.6 (13.8 – 275)	87.1	57.2
Entire watershed	49,650	446 (76.3 –1274)	88.4	64.1
Modeled system[c]	49,720	--	--	64.9

[a]Becher et al., 2001
[b] USGS water resources data for Iowa, http://nwis.waterdata.usgs.gov/ia/nwis/
[c] based on sum or acreage-weighted average of county data

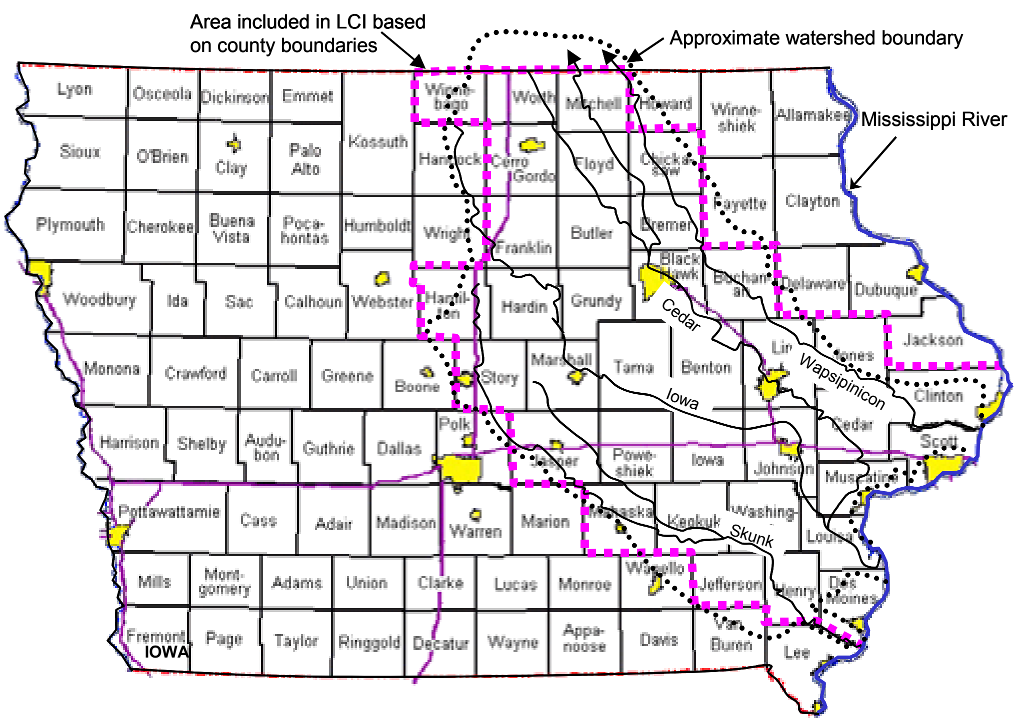


Figure 9. Map of Iowa identifies the watershed for the four major river systems in Eastern Iowa and the corresponding area included in the LCI based on county boundaries. (Watershed definition based on Iowa DNR map, http://www.iowadnr.com/water/tmdlwqa/wqa/303d/1998/map1.gif)

The geology, hydrology and land use practices in this region are described extensively by Becher et al., (1999; 2001). Overall, approximately 64% of the region is used for row crop production. This is mostly dedicated to corn-soybean rotations (C-S), although some farms, especially in the northeast and east central regions, practice corn-corn-soybean rotations (Brenner et al., 2001). This accounts for the slightly greater row crop land dedicated to corn (59%) every year versus soybeans (41%).

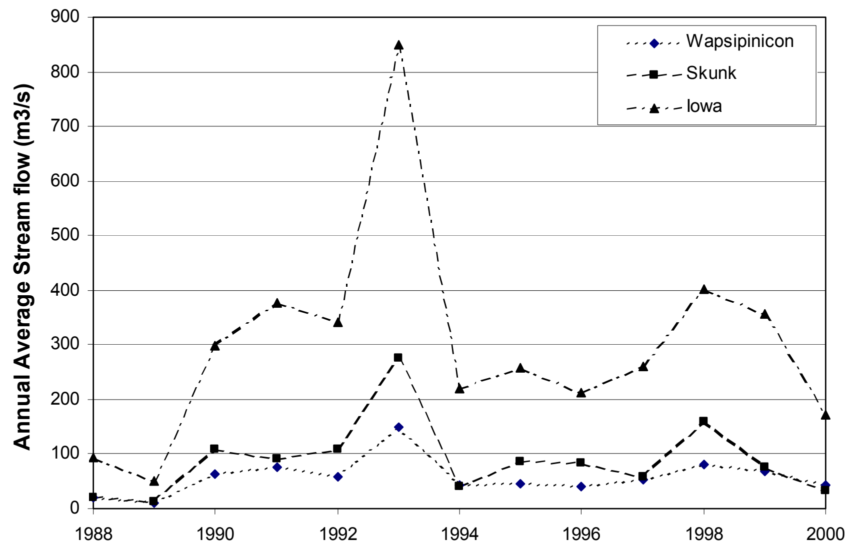
There are four major landforms in eastern Iowa (Becher et al. 2000). The Des Moines lobe in north central Iowa was formed by glaciation and is characterized by very low topographical relief and thick loamy till soils. The Iowa surface in NE Iowa also has low relief and loamy till, but with a thin cover of loess on ridges or alluvium near streams. A portion of this region is defined as the Iowa karst subsection with typical sinkhole topography. Southeastern Iowa is defined by the southern Iowa Drift Plain. It has greater relief and steep rolling terrain with flat tabular divides between the hills. The soil is composed of glacial loess. Because of the low relief and till soils in the northern two-thirds of the study area, extensive tile drainage is required to maintain proper soil moisture for crop health.

The lifecycle boundaries considered here include cradle-to-farm gate flows, resulting in the production of corn, soybeans or corn stover that could later be used for bio-based products or fuels. As shown in Figure 11, the system considers:

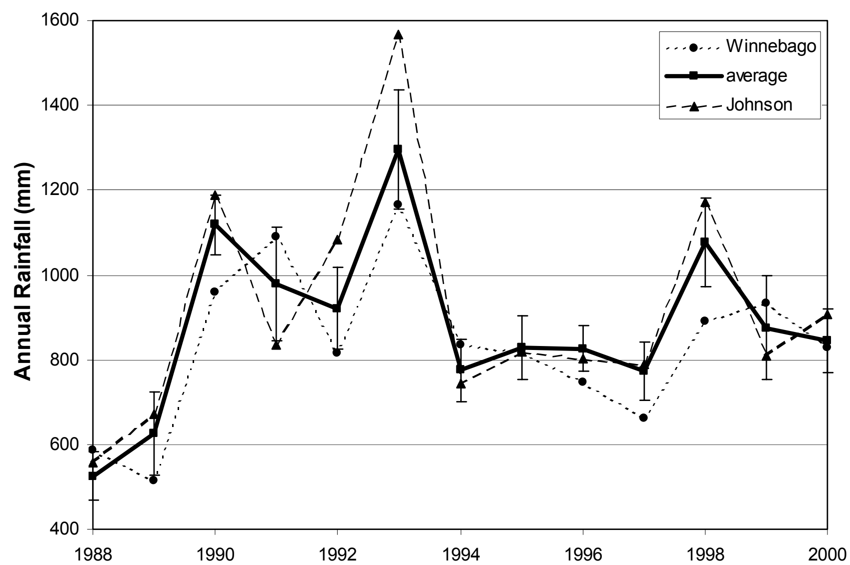
- Land area involved with corn or soy production
- All nutrients (N, K, P) used to improve crop yields
- Nutrient flows (NO_3, NO_x, N_2O, TN as nitrogen, TP) to water and air compartments (as applicable)
- N and P flow to the Mississippi River and the Gulf of Mexico
- Carbon flows related to plant uptake and harvest, energy consumption
- Diesel fuel, gasoline and motor oil used on the farm for crop planting and harvesting
- Upstream generation of fertilizers and energy along with their LC nutrient and energy related emissions
- Eutrophication, acidification and greenhouse gas impacts

The system does not include flows related to

- Other farm related energy uses
- Farm or upstream processing equipment manufacture and maintenance
- Other agrichemicals used on the farm (e.g., herbicides)
- Other LC impacts (most notably – toxicity related to nitrates, fossil fuel emissions, or herbicides)



a)



b)

Figure 10. Annual average stream flow (a) for Eastern Iowa watersheds and (b) regional average and example county precipitation values
(USGS water resources data for Iowa, http://nwis.waterdata.usgs.gov/ia/nwis/ ; Spatial Climate Analysis Service http://www.ocs.orst.edu/prism/)

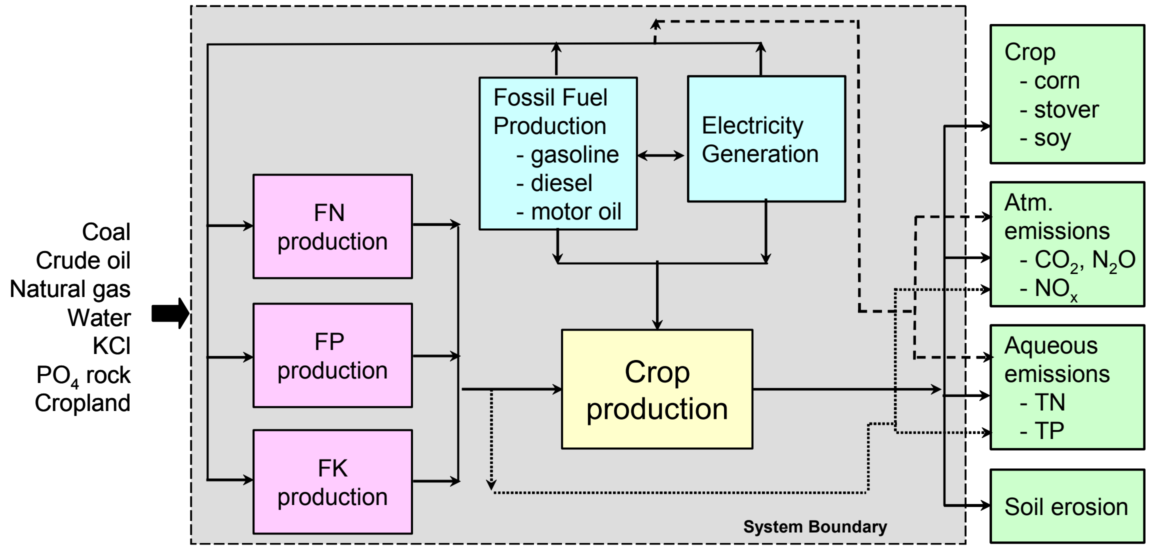


Figure 11. System boundary for cradle-to-gate life cycle assessment for corn and soybean production

Stover Removal Model

The stover harvest rate was defined based on residue production in a given year and the minimum stover residue required limiting erosion to an acceptable level. Some residue is required to remain on the field to prevent excessive erosion. Minimum stover residue requirements for conventional, mulch and no till practices were calculated by Richard Nelson for an NREL report (Nelson 2004) for C-S rotations and for the stover LCA (Sheehan et al., 2002). Nelson used standard erosion models (RUSLE) with long-term average values for rainfall and grain yields to determine required remaining residue. His values for minimum residue requirements do not integrate variability among years associated with the weather.

For the C-C system, the amount of residue that can be harvested in each county was estimated as:

$$H_{st,harv} = H_{res,c} - H_{res,min} \qquad [7]$$

where, $H_{st,harv}$ is the yield of harvestable stover (kg/ha corn) and $H_{res,min}$ is the crop residue that needs to remain on the field to limit erosion to a tolerable level. The amount of residue generated is defined as $H_{res,c} = H_c \cdot HI_c$,where H_c is the corn grain yield and HI_c is the harvest index for corn. In the study reported here, stover harvest values vary among years due to the use of annual H_c values for each of the 13 years considered.

Determining the quantity of stover that can be harvested in a C-S rotation is more complicated (Nelson, 2004). In this case, the minimum amount of residue remaining on the field must be compared to a two-year average of crop residue generated during a complete C-S rotation and actual amounts for each of the crops. The average residue generated ($H_{res,avg}$) was defined as:

$$H_{res,\,avg} = H_c \cdot HI_c + H_{res,soy,t-1} \cdot HI_{soy} \qquad [8]$$

where $H_{res,soy,t-1}$ is the soy residue generated in the previous year.

For the C-S system, the amount of stover that could be harvested in a given year was determine by the following algorithm (Nelson, 2004):

- If both the corn and soy residue amounts exceed the minimum required remaining residue over the two-year period, the maximum amount of stover can be harvested:

$$\text{IF } H_{res,c} > H_{res,min} \text{ AND } H_{res,soy,t-1} > H_{res,min} \text{ THEN } H_{st,harv} = H_{res,c} - H_{res,min} \quad [9]$$

- If the amount of stover generated is less than the required amount remaining, no stover can be harvested:

$$\text{IF } H_{res,c} < H_{res,min} \text{ THEN } H_{st,harv} = 0 \quad [10]$$

- If the *average* amount of residue generated over a two-year period is less than the required amount remaining, no stover can be harvested:

$$\text{IF } H_{res,\,avg} < H_{res,min} \text{ THEN } H_{st,harv} = 0 \quad [11]$$

- If the *average* residue generated is greater than the allowable, but the soy residue is less than the allowable, some stover can be collected, but at a yield less than the maximum:

$$\text{IF } H_{res,avg} > H_{res,min} \text{ AND } H_{res,soy,t-1} < H_{res,min} \text{ THEN } H_{st,harv} = (H_{res,avg} - H_{res,min}) \cdot 2 \quad [12]$$

Data required for estimating the stover yields include: corn and soybean yields for each county and each year (USDA NASS); harvest indices (Gupta, 1978); and required minimum residue values for each county, which have been calculated and provided by Richard Nelson (2004; Sheehan et al., 2002).

Nitrogen Model Development and Input Data

An empirical approach is employed here that defines individual flows of nitrogen through the system as fractions of the known nitrogen inputs to the overall system (field, atmosphere, water). Key species of interest include ammonia/ammonium, nitrate (NO_3^-) (small quantities of nitrite present are included in this term), nitrous oxide (N_2O) and nitrogen oxides (NO_x, mostly as NO). All flows are presented in terms of their nitrogen mass (kg-N). All input flows, rates and fractions are defined and quantified in Tables A-1 through A-4. Symbols are shown on Figure 3.

The approach used here is most similar to that employed by deVries et al. (2003), and includes substantial ideas based on the work of Burkart and James (1999), Galloway et al. (2003), IPCC (1996), Meisinger and Randall (1991), and Goolsby et al. (1999). The model tracks individual nitrogen species and their particular fate rather than lumping species and processes together before fractioning to define the distribution of the nitrogen in various compartments. The deVries et al. (2003) work applies fractions suitable for the Netherlands and all types of land use. The model developed here uses a range of literature sources and field data to better quantify the flows and fractions of flows that are more applicable to corn and soybean production in eastern Iowa.

Anthropogenic reactive nitrogen is applied directly as chemical or animal fertilizer and indirectly through atmospheric deposition. We assume that animal manures are infrequently used in Eastern Iowa and that anhydrous ammonia is used most widely as a commercial inorganic fertilizer (Hanna et al., 2002). Nitrogen available for crops and leaching is also supplied by nitrogen fixation by soybean plants and release of nitrate from crop residues and soil organic matter (SOM) during mineralization.

Except for a small fraction of the ammonia fertilizers applied that are lost by volatilization, most of the remainder is converted through the microbial nitrification process into nitrate. Small, but important quantities of NO, N_2O and NO_2 are produced in the nitrification process. The nitrate produced is predominantly available for plant uptake. It is also stored in SOM, leached into ground and surface water, and anaerobically denitrified to the unreactive form of nitrogen – N_2, with N_2O and NO generated as by-products. Although most nitrates are eventually denitrified, they remain in the soil and aqueous systems long enough to cause deleterious ecosystem and human health impacts.

The flows included in this model include:

Key Input flows

- Fertilizer application
- Atmospheric deposition (wet and dry)
- Mineralization of SOM
- Mineralization of remaining crop residues
- N_2 fixation

Key transformation and output flows

- Ammonia volatilization (fast)
- Leaching to SW (through tiles and base flow expressed as TN – including soluble and particulate forms)
- Leaching to groundwater
- Nitrification ($NH_3 \rightarrow NO_3^-$, including losses as NO and N_2O)
- Denitrification ($NO_3^- \rightarrow N_2$, including losses as NO and N_2O)
- Crop harvest (including stover)
- Nitrogen immobilization within SOM
- Plant senescencing

Ammonia Sources

The total ammonia load (L_{NH3}, kg yr^{-1}) added to this system is the sum of fertilizer use (L_f), ammonification of organic nitrogen ($L_{soil\ min} + L_{crop\ min}$), and atmospheric deposition ($L_{atm,NH3}$):

$$L_{NH3} = L_f + L_{atm,NH3} + L_{soil\ min} + L_{crop\ min} \qquad [\ 13]$$

The fertilizer load includes that applied to corn ($L_{f,c}$), soy ($L_{f,soy}$) and the additional fertilizer required to replace the nitrogen lost from the system with the stover harvest the previous year (t-1).

$$L_f = A_{c,p} \cdot N_{f,c} \cdot f_{f,c} + A_{c,H} \cdot \left(\frac{H_{st,har}^{(t-1)} \cdot f_{st,N}}{1 - f_{vol}} \right) + A_{soy,p} \cdot N_{f,soy} \cdot f_{f,soy} \qquad [\ 14]$$

Appropriate historical values for the area planted ($A_{i,p}$) are multiplied by the application rates ($N_{f,i}$) and fraction of acres fertilized ($f_{f,i}$) to quantify the loads due to fertilization. Subscripts c, st, and soy refer to corn, stover and soybeans, respectively. The middle term accounts for fertilizer nitrogen that must be added to replace the nitrogen removed with the stover harvest the previous year ($H_{st,harv}^{(t-1)}$) (see eqn. 7-12). The fraction of nitrogen in the stover ($f_{st,N}$) is required for this calculation. As suggested by IPCC, FN needs are adjusted to account for the fraction that is rapidly lost via volatilization (f_{vol}).

Wet deposition can be defined by rainfall (Q_{rain}) and NH_3 concentration in rain (C_{NH3}). A_{ws} is the total area of the watershed. Dry deposition data are scarcer so this term is typically estimated as a fraction of the wet deposition (f_{DD}).

$$L_{atm,NH_3} = A_{ws} \cdot (1 + f_{DD,NH3}) \cdot (Q_{rain} \cdot C_{NH_3}) \cdot 10^{-5} \qquad [\ 15]$$

Concentration data are summarized by the National Atmospheric Deposition Program for wet deposition are available at three sites around the eastern Iowa watershed system (NADP, http://nadp.sws.uiuc.edu). These three sites do not include dry deposition measurements though. The closest site with dry deposition data is in east central Illinois (Bondville). Goolsby et al. (1999) summarize average data for this site for 1992 – 1993. They found $f_{DD,NH3}$= 0.11, with higher values further east. and $f_{DD,NO3}$= 0.41

Organic matter in the system, which includes both SOM and crop residues, can undergo ammonification reaction (Canter, 1997). As described below, this ammonia is then typically quickly mineralized to nitrate.

The extent of these reactions depends on the quantity of organic matter, how much of it is mineralized in a given year, and its nitrogen content. Burkart and James (1999) quantify soil mineralization as:

$$L_{soil\ min} = (A_{c,p} + A_{soy,p}) \cdot N_{soil,min} \qquad [16a]$$

$$\text{where} \quad N_{soil,min} = \rho_b \cdot V_{soil} \cdot f_{om} \cdot f_{om,N} \cdot f_{min} \qquad [16b]$$

where V_{soil} is the volume of soil in the top 30 cm, ρ_b is the bulk density (~1480 kg/m^3), f_{om} quantifies the fraction of organic matter in the soil, $f_{om,N}$ is the fraction of nitrogen in the SOM (3%), and f_{min} is the fraction of SOM that is mineralized in a given year, typically 1-3%. The fraction mineralized varies with cultivation, with lower values for no-till soils (Schepers and Moiser, 1991). Using this approach, Goolsby et al. (1999) estimated >50 kg/ha of nitrogen are released annually in Iowa and other corn belt states due to soil mineralization. Gentry et al. (2001) measured annual mineralization rates of 72 to 112 kg-N/ha in a high organic content soil in Illinois (f_{om}=0.045 – 0.05), with the variation attributed to the previous year's crop (low values for corn, high values for soy). Calibrating eqn. [16b] to these data (f_{om}=0.045) provides estimates of $f_{min,c(t-1)}$ = 0.014 and $f_{min,s(t-1)}$ = 0.02, where the term (t-1) in the subscript indicates that corn or soy was grown in the previous year.

In the absence of such data, other researchers have used a range of approaches, including an assumption that this soil mineralization is negligible for all but soils with very high SOM (e.g., peat, deVries et al., 2003).

We assume that all crop residues that are left on the field from the previous year (t-1) are mineralized. This includes all of the soy straw and the corn stover that is not harvested. The soy straw actually degrades very quickly after harvest, and would really be released in the modeled, not following year (Gentry et al., 2001). Corn on the other hand, degrades more slowly, with 10-20% remaining after 2 years (Wilhelm et al., 2004).

$$L_{crop\ min} = A^{t-1}_{c,H} \cdot \left(H^{t-1}_{c} \cdot HI_c - H^{t-1}_{st,harv}\right) \cdot f_{st,N} + A^{t-1}_{soy,H} \cdot H^{t-1}_{soy} \cdot HI_{soy} \cdot f_{soy\text{-}st,N} \qquad [\ 17]$$

where the area of concern here is the harvested area ($A_{i,H}$), not planted area, and $f_{i,N}$ is the mass fraction of nitrogen in the plant material.

Ammonia Sinks

Ammonia exists as a gas and can readily volatilize quickly after fertilizer is applied. The extent of this loss is quantified as:

$$L_{vol} = L_f \cdot f_{vol} \qquad [\ 18]$$

The fraction that is volatilized (f_{vol}) can range from 0-50% depending on the chemical formulation of the inorganic nitrogen fertilizer, the manner in which it is applied, and the soil pH. The injection of anhydrous ammonia results in the lowest loss rates (f_{vol} = 0.0 – 0.05) (Meisinger and Randall, 1991). We assume that ammonia entering the system as atmospheric deposition or via ammonification does not volatilize.

Ammonia gas is also lost directly from plant leaves at the end of the growing season. Plant senescencing can contribute $N_{senes,c}$ ~ 50 kg/ha/yr for corn and $N_{senes,soy}$ ~ 45 kg/ha/yr for soy (Burkart and James, 1999). Goolsby et al. (1999) used $N_{senes,c}$ = 60 kg/ha/yr for corn and the same value for soy as Burkart and James. The total flow can be estimated as:

$$L_{senes} = A_{c,H} \cdot N_{senes,c} + A_{soy,H} \cdot N_{senes,soy} \qquad [\ 19]$$

Data are insufficient to estimate this flow on a more appropriate basis of plant yield, climate or other factors (Bouwman et al., 1997).

Most of the ammonia applied or deposited on cropland stays within the system but is transformed by nitrification, primarily into nitrate.

$$L_{ni} = (L_f \cdot (1 - f_{vol}) + L_{atm,NH3} + L_{soil\ min} + L_{crop\ min}) \cdot f_{ni} \qquad [20]$$

where f_{ni} is the fraction of available ammonia that is nitrified. This assumes that volatilization occurs quickly so that that fraction that volatilizes is not available for nitrification. The small fraction of remaining ammonia, which is incorporated into plant tissue and SOM, is not quantified.

Nitrate Sources

Most of the nitrate in the system is transformed from ammonia through the nitrification process (eqn. [20]), although atmospheric deposition also contributes to the total amount of nitrate that is available for plant growth and leaching.

$$L_{NO3} = L_{ni} \cdot (1 - f_{ni,NO} - f_{ni,N2O}) + L_{atm,NO3} \qquad [21]$$

The ammonia loss by nitrification (L_{ni}) is adjusted to quantify the amount of nitrate generated by recognizing that some is lost as NO and N_2O. Atmospheric deposition ($L_{atm,NO3}$) provides additional sources in both dry and wet forms. These flows are calculated in the same manner as described for ammonia (Eqn. [15]).

$$L_{atm,N03} = A_{ws} \cdot (1+f_{DD,NO3}) \cdot (Q_{rain} \cdot C_{NO3}) \cdot 10^{-5} \qquad [22]$$

The analysis by Goolsby et al. (1999) provides an estimate of $f_{DD,NO3} = 0.41$ for the closest site (east central IL) with both wet and dry deposition data. Concentration data are available for three NADP sites surrounding the Eastern Iowa watershed system.

Nitrate Sinks

Most of the nitrate is taken into the plant mass and subsequently leaves the system as a harvested crop (L_{harv}) or remains as residue on the soil (L_{res}), to be recycled the following year as a nitrate source (eqn. [17]). Significant amounts are lost, however, through leaching (L_{sw}, L_{gw}), immobilization as organic nitrogen (L_{imm}), and denitrification (L_{de}).

Crop Harvest

Agricultural data quantifying crop yields and nitrogen content in plants can be used to define the nitrogen leaving the system with the plant matter.

$$L_{harv} = A_{c,H} \cdot (H_c \cdot f_{c,N} + H_{st,harv} \cdot f_{st,N}) + A_{soy,H} \cdot H_{soy} \cdot f_{soy,N} \qquad [23]$$

Where H indicates the mass harvested, $f_{i,N}$ is the mass fraction of nitrogen in the material harvested.

The nitrate bound in crop residues is defined in a similar manner:

$$L_{res} = A_{c,H} \cdot (H_c \cdot HI_c - H_{st,harv}) \cdot f_{st,N} + A_{soy,H} \cdot H_{soy} \cdot HI_{soy} \cdot f_{soy\text{-}st,N} = L_{crop\ min\ (t=t+1\ yr)} \qquad [24]$$

This assumes that the minimum required stover and all of the soy straw is left on the field. Note that this equation is identical to the crop residue mineralization term. The mineralization is assumed, however, to occur in the following year.

Leaching

Nitrate species are very soluble and are often found in water leaching to surface or groundwater. The extent of leaching depends on both the total flow of water through the system and the amount of nitrate available for leaching. Numerous studies assume that the amount of nitrogen leached can be estimated directly as a fraction of the fertilizer nitrogen (Table 6) or net anthropogenic nitrogen[1] applied (Howarth et al., 1996). However, this does not capture the substantial variation in the leaching during years of light versus heavy rainfall that has been observed at several field sites (Lucey and Goolsby,1993; Puckett et al., 1999; Weed and Kanwar, 1996; Jaynes et al., 1999; Bakhsh et al. 2002).

Table 6. Range of Estimates - Percent of FN that Leaches with Water

Source	Average % FN that leaches	Comments
GREET (Wang, 1999)	24%	Used in LCA for subsequent denitrification and N_2O emission calculations for transportation fuels
IPCC (1996)	30%	Global average value used for subsequent denitrification and N_2O emission calculations
Puckett et al. (1999)	27%	Measured - Leached into groundwater in highly permeable outwash aquifer (west MN) (assumed no runoff)
Lucey and Goolsby (1993)	25% (1.5 – 64%)	Measured values at downstream location. Based on cumulative load in Raccoon River (central IA) over 11 years versus cumulative FN used. Range in %FN leached consistent to range of annual rainfall. (Note – the down stream collection site for this study would lump in-stream denitrification into this fraction, concentrations at field would be higher)
Based on data from Weed and Kanwar (1996) [a]	36% (8 - 59%)	Measured values – 3 yr. field scale study. Nashua IA
Based on data in Bakhsh et al. (2002)	26% (5 – 76%)	Measured values – 6 yr. field scale study. Nashua IA (NE IA). Rainfall rates most significant variable.
Based on data in Jaynes et al. and Hatfield et al. (1999)[a]	42% (9 – 85%)	Measured - Walnut Creek watershed study, central IA, average of several drainage-scale systems
Based on data summarized by Bakhsh et al. (2004)	47%	Overall 6 year cumulative average fraction FN leached from single C-S rotation field in Walnut Creek watershed, central IA, (1992 – 1997)

[a] data used for calibration of leaching model presented in Figure 12.

Figure 12 presents the analysis of leaching data for agricultural research stations in Iowa and nearby regions to provide a means of estimating FN leaching in eastern Iowa. These data quantify the fraction of fertilizer nitrogen applied to the field that was lost with tile drainage as a function of the annual rainfall (mm). FN is used as a surrogate variable to quantify all nitrate leaching sources associated with row crop agriculture. In reality, atmospheric deposition, nitrogen fixation and soil mineralization also contribute to the total measured nitrate loads in the drains. The inclusion of terms to quantify these nitrogen sources as well as FN was attempted. The large uncertainties, especially with soil mineralization, prevented the development of meaningful relationships.

The regression equation for corn-soybean fields includes both field scale (Nashua IA) and drainage scale (Walnut Creek) data with a range of tillage (CP- chisel plow, NT – no till) and nitrogen application practices (SA – single fall application, LS – late spring). For the field data, the fraction of fertilizer nitrogen leached was estimated over a two-acre system that assumed equal acreage in soy and corn.

[1] Net anthropogenic input: NAI = fertilizer + crop N fixation + food/feed imports – food exports

Fertilizer was only applied over 50% of this system, but nitrate leaching was similar for both soy and corn farmland. This indicates that there is residual FN remaining after the corn harvest and/or that other sources of nitrate contribute substantially to nitrate leaching during the soy rotation. One regression equation for C-S was determined for these two different sets of data even though there are statistically significant differences in their individual regression equations. Significantly higher leaching rates were observed at the larger scale Walnut Creek site. The cause of differences among these data sets can be attributed to geographical differences (hydrology, geology) or differences in the scale of measurement and, therefore, in the mechanisms that affect the nitrate fate.

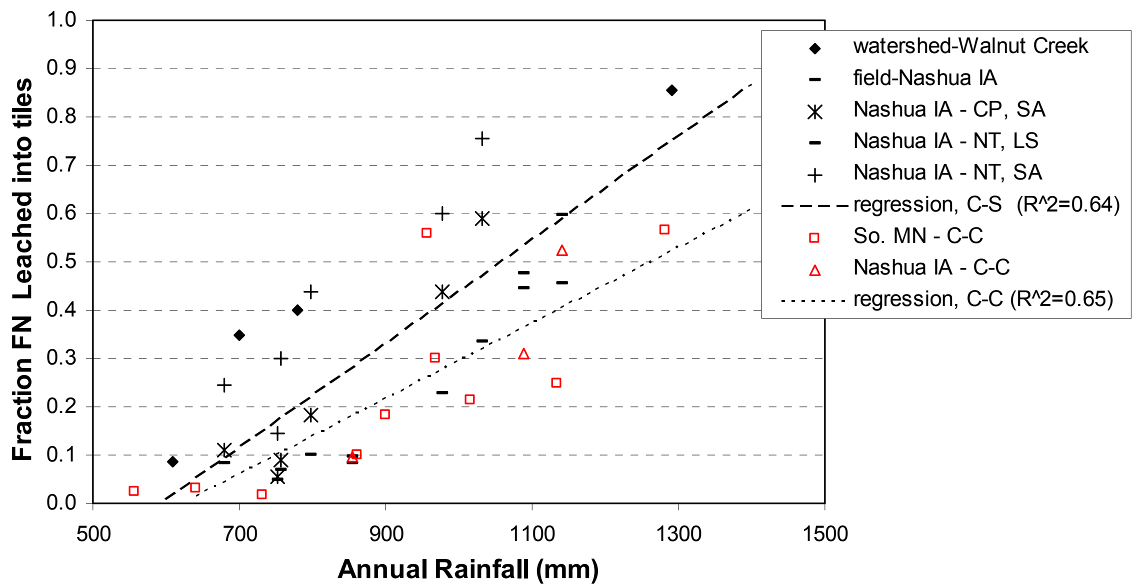


Figure 12. Fraction of nitrogen leached as a linear function of annual rainfall.
Data from Walnut Creek (Jaynes et al., 1999; Hatfield et al., 1999), Nashua IA (Weed and Kumar, 1996; Bjorneberg et al., 1996; Bakhsh et al., 2002), and southern MN (Randall and Iragavarapu, 1995).

Data for continuous corn operations were only available for field-scale systems. The regression analysis suggests that a lower ***fraction*** is leached from C-C than C-S (this does not imply that less total mass of TN is leached from C-C rotations). It is acknowledged that the primary difference here is the type of data available between these two crop rotations. Table 7 also includes a regression equation for a hillier and naturally drained field site in western Iowa (Steinheimer et al., 1998) for comparison.

The right hand column in Table 7 compares the percentage FN that leaches based on the regression equations applied at an average rainfall rate for Cedar Rapids in eastern IA (917 mm; http://mesonet.agron.iastate.edu/climodat/index.phtml). The variability in these values suggests a wide range of estimated fractional losses due to differences in geographic location, crop rotation and/or tillage. Analysis of data from Weed and Kanwar (1996) at a single site, however, shows little differences among the various crop rotations and tillage methods used compared with differences among the three years studied, which included a range of precipitation rates. Thus, much of the differences seen among the data and between the regression lines in Figure 12 can be attributed to the scale of the measurement and/or geographic location; it is difficult to attribute these differences to just differences in the C-C versus C-S rotations. Care must be used to not apply leaching rates and or regression equations such as shown in Figure 12 over a broad geographic region. Differences in geology and hydrology could result in much different leaching rates. Data pertinent to the regional characteristics could be acquired and analyzed in a fashion similar to what was done here to apply this approach to other areas.

Table 7. Regression Equations Defining the Fraction of Fertilizer Nitrogen that Leaches into Tiles and Drains

System	Slope[a] (± 95% C.I.) (mm^{-1})	Intercept (± 95% C.I.) (--)	R^2	% FN leached[b]
C-S, field and watershed scale, eastern IA, range of tillage practices	0.00103 (± 0.00029)	-0.480 (± 0.255)	0.643	46%
C-C, chisel plow, E. IA	0.000779 (± 0.000378)	-0.483 (± 0.361)	0.652	23%
C-C, chisel plow, W. IA	0.000675 (± 0.000473)	-0.222 (± 0.346)	0.536	40%

[a] $f_{NO3,SW}$ = slope * rainfall (mm) + intercept
[b] using long-term average annual rainfall (917 mm) at Cedar Rapids, IA

The simple models presented in Figure 12 capture some of the annual variability in nitrogen leaching rates and thus are an improvement over those that use a single fraction of the FN to estimate the amount leached. This approach, however, is still subject to a large amount of uncertainty and fails to incorporate variability that stems from longer-term accumulation and release of soil nitrogen that occurs with a series of dry or wet years. A more mechanistic model would have to be used to incorporate these variables.

Based on the field data and analysis presented above, it is assumed that the amount of nitrogen leached to surface water can be quantified as:

$$L_{SW} = L_f \cdot f_{NO3,SW} \qquad [25]$$

where $f_{NO3,SW}$ is defined by a linear function of the annual rainfall.

There is significantly less data to quantify the flux of nitrate into groundwater systems below the depth of the tiles or drains. In the absence of sufficient data to estimate the variation in this flow with rainfall and percolation rates, a straight fraction of the net nitrate input can be used to estimate the flow of nitrate to the groundwater. The net anthropogenic nitrate inputs here are approximated as the anthropogenic inputs minus the crop harvest.

$$L_{GW} = (L_f + L_{atm,NO3} - L_{harv}) \cdot f_{NO3,GW} \qquad [26]$$

Based on an analysis of nitrogen cycling over the intensively cultivated areas within the United States, Howarth et al. (1996) estimate that approximately 3% of net anthropogenic nitrogen leaches to groundwater. In a more geographically applicable analysis, David and Gentry (2000) assumed that no nitrate enters and remains in groundwater in Illinois. The extensive use of tile drains in Illinois helps to substantiate this assumption, although nitrate contamination of groundwater systems has been observed. Thus, although this sink might be small from an overall mass balance perspective, it could be significant from a resource degradation standpoint. In fact, application of eqn. [25] to eastern Iowa data showed that in some years there were negative amounts of net available nitrogen as defined in this way. Thus, groundwater leaching is considered negligible here from a mass balance perspective. Higher fractions might be observed for western Iowa and Nebraska where the hydrology is influenced by irrigation and sandier soils than in E. IA and IL.

Denitrification

In an anaerobic environment and in the presence of sufficient organic matter, nitrates are denitrified to N_2, with NO and N_2O generated as by-products (Canter 1997). Denitrification can occur in the soil, groundwater and surface waters. Rates are highest in poorly drained soils and slowly moving and/or shallow surface water bodies (e.g., shallow lakes or wetlands), although some denitrification also occurs in shallow streams as well. The total extent of denitrification can be defined as the sum of losses in each of the environmental compartments:

$$L_{de} = L_{de,s} + L_{de,gw} + L_{de,dr} + L_{de,ri} + L_{de,gu} \quad [27]$$

where subscripts s, gw, dr, ri and gu refer to the soil solution, deeper groundwater, drainage system, rivers, and Gulf of Mexico, respectively. Only the first three terms are important when considering the nitrate losses within the geographic boundary of Eastern Iowa. The first four terms are important for determining the contributions of fertilizer use in Iowa on the hypoxia condition in the Gulf of Mexico, and all the terms are important for quantifying NO and N_2O emissions associated with the agricultural activity.

Unfortunately, most researchers estimate only the overall fraction of applied reactive nitrogen, typically fertilizer + atmospheric deposition, that is denitrified (Howarth et al., 1996; Meisinger and Randall, 1991):

$$L_{de} = f_{de} \cdot (L_{atm,NO3} + L_f) \quad [28]$$

where the fraction denitrified (f_{de}) is generally defined between 6 and 25%. This approach reduces the value of knowing the amount on nitrate in the system at intermediate steps as it is transported to a coastal system.

Most denitrification reactions occur in the soil since this represents the largest pool of nitrogen. The fraction of nitrate in the soils that denitrifies can range from 2-20% (Meisinger and Randall, 1991, Burkart and James, 1999) of the available nitrate. Higher fractions are expected in more poorly drained and/or organic rich soils. For the average organic content of soils in Iowa (f_{om}=0.035), Goolsby et al. (1999) used $f_{de,s}$=0.20 applied to available fertilizer nitrogen (adjusted for volatile losses).

DeVries et al. (2003) used a mass balance approach to estimate the nitrogen remaining in soil which is available for denitrification:

$$L_{de,s} = f_{de,s} \cdot (L_{atm,NO3} + L_f \cdot (1-f_{vol}) + L_{fix} - (L_{imm} + L_{harv} + L_{res})) \quad [29a]$$

where the $f_{de,s}$ varies with soil type and moisture. For Loess soils, deVries et al. (2003) define $f_{de,s} \sim 0.40 - 0.80$ for moist soils and 0.60-0.95 for wet soils. These fractions are higher than others have used, but they are applied to a smaller mass of nitrate remaining in the soil. Thus, use of these emission factors should be carefully coordinated with the correct nitrogen mass.

Similar equations can be developed for other compartments:

$$L_{de,gw} = f_{de,gw} \cdot L_{gw} \quad [29b]$$

$$L_{de,dr} = f_{de,dr} \cdot L_{sw} \quad [29c]$$

$$L_{de,ri} = f_{de,ri} \cdot (L_{sw} - L_{de,dr}) \quad [29d]$$

$$L_{de,Gu} = f_{de,Gu} \cdot (L_{sw} - L_{de,dr} - L_{de,ri}) \quad [29e]$$

deVries et al. (2003) assumed $f_{de,gw}$ and $f_{de,dr}$ were equal to the value of $f_{de,s}$ for wet soils and $f_{de,ri} \sim 0.6$-1.0 for a set of rivers in the Netherlands. Galloway et al. (2003) cite a value for $f_{de,gw} = 0.4$ for So. MN and general fractions $f_{de,ri} \sim 0.3 - 0.7$, $f_{de,est} = 0.1 - 0.8$ (estuaries), and $f_{deGu} \sim >0.8$. An additional term for estuaries was defined by Galloway since this is an important point in the surface water flow where the residence times decrease and the denitrification potential increases. The data used here to generate regression equations to estimate L_{sw} were generally collected at the point where the tile or drain discharged to a creek or river. Thus, any denitrification that occurred in the drains is already accounted for by this data, so $f_{de,dr}$ is set to zero since it is already lumped into the calculation of L_{sw}.

Like leaching, denitrification factors are known to depend on environmental conditions and should be adjusted to account for annual variability in climatic conditions. Smith et al. (1997) define first order denitrification rates for US river stretches as a function of the annual flow rate. They define the lowest decay rates for rivers with annual flow rates > 283 m^3/s (>10,000 ft^3/s) and the highest decay rates for rivers with annual flow rates <28.3 m^3/s (>1,000 ft^3/s). Although applying these decay constants would require sophisticated water quality models such as the SPARROW model employed by Smith et al (1997), we can use these ranges to choose reasonable initial values of $f_{de,ri}$ for the different watersheds depending on their actual annual flow rates. This assumes that the overall nature of the watershed systems are similar, including the number, depth and length of reaches contributing to the river and the overall residence time. The major watersheds in Eastern Iowa have annual discharges ranging from <28.3 m^3/s to >283 m^3/s (Table 8). The Mississippi River at Clinton Iowa has an annual flow rate generally greater than 1000 m^3/s. It is typically assumed that there is no denitrification between entry of the nitrate into the Mississippi River and its discharge into the coastal waters of Louisiana (McIsaac et al. 2002). Others have completed experimental investigations to confirm that nitrogen losses in the Mississippi River are low (Battaglin et al. 2001; Richardson et al., 2004).

Table 8. Denitrification Factors for Surface Water

Watershed	Years between 1988 and 2000 when watershed annual flow rate within specified range		
	<28.3 m^3/s	28.3 – 283 m^3/s	> 283 m^3/s
Wapsipinicon	'88 – '89	'90 – '00	--
Iowa (including Cedar)	--	'88 – '89 '94 –'97	'90 – '93 '98 – '00
Skunk	'88 – '89	'90 – '00	--
Initially assumed range[a] for $f_{de,ri}$	0.5 – 0.7	0.4 – 0.6	0.3 – 0.5

[a] based on overall range provided by Galloway et al. (2003) and the specific river flow rates

Immobilization

Nitrates can be slowly converted to organic nitrogen compounds and immobilized in the SOM. These compounds can be mineralized at a later date releasing the nitrogen for leaching or plant uptake. Over a period of several years, the soil nitrogen content has been relatively stable, suggesting that the immobilization and denitrification terms cancel each other out. On a year-to-year basis, however, there can be substantial changes in the soil nitrogen content due to reduced plant uptake during dry or years or increased leaching during wet years. Goolsby et al. (1999) suggest that the amount of nitrogen immobilized be estimated as a fraction of the anthropogenic nitrogen inputs:

$$L_{imm} = L_f \cdot f_{f,imm} + L_{atm,NO3} \cdot f_{dep,imm} \qquad [30]$$

The fractions $f_{f,imm}$ and $f_{dep,imm}$ were both set to 0.4 by Goolsby et al. (1999), with an estimated 25-50% uncertainty. There is very limited data to improve these estimates based on physical measurements or mechanistic modeling.

Other Nitrogen Flows

Nitrogen Fixation

Soybeans and other legumes can directly convert nitrogen from the atmosphere into nitrogen required for plant matter. Meisinger and Randall (1991) suggest that the nitrogen fixation rates can be estimated as a fraction of the nitrogen content in the plant material:

$$L_{fix} = A_{soy,p} \cdot H_{soy} \cdot (f_{soy,N} + HI_{soy} \cdot f_{soy\text{-}res,N}) \cdot f_{fix,N} \qquad [31]$$

where HI is the harvest index for soy and $f_{fix,N}$ is the fraction of the plant nitrogen content that is attributable to nitrogen fixation. Plants tend to use readily available soil nitrogen first and then fix additional nitrogen as needed (Gentry et al., 1998). $f_{fix,N}$ can range from 0.5 to 0.8 and decreases with increasing availability of nitrogen from commercial fertilizer or soil reserves (Meisinger and Randall, 1991).

Alternatively, a simpler approach is based solely on the acreage of soy and other legumes (Meisinger and Randall, 1991, Goolsby et al., 1999):

$$L_{fix} = A_{soy,p} \cdot N_{fix,N} \qquad [32]$$

where $N_{fix,N}$ is the rate at which nitrogen is fixed on soy cropland. Jordan and Weller (1996) provide an estimate for this parameter $N_{fix,N}$ = 78 kg/ha with a range of 15-310 kg/ha. Applying typical value for soy yield in Iowa (2.74 Mg/ha) and the fractions quantifying nitrogen content of soy and soy straw (Table A-3), eqn. [31] predicts a nitrogen fixation rate of 98 kg/ha soy cropland. This is within the range of values for $N_{fix,N}$ summarized by Goolsby et al. (1999).

Erosion

The mass of soil eroded ($M_{soil,er}$) varies with soil type, agricultural practices, land use and rainfall intensity. Nelson (2002, 2004) used the generally accepted RUSLE soil erosion model to estimate soil erosion for each soil type in Iowa as a function of crop rotation and tillage practices for long-term average rainfall data. An area-weighted average of soil types and their associated erosion estimates were used here to determine the average mass of soil lost from each county in E. Iowa for conventional till practices.

Not all eroded soil gets to waterways and degrades water quality. The sediment delivery ratio ($f_{er,sed}$) defines that fraction of the total eroded soil that moves into waterways. Sheehan et al. (1998) estimated average sediment delivery ratios for major soybean producing states based on the work of Ribaudo (1989). They estimate $f_{er,sed}$=0.512 for Iowa.

Nitrogen, mostly in the form or organic-N in the SOM is transported to surface water bodies with eroded soil. This load is estimated as:

$$L_{er} = (A_{c,p} + A_{soy,p}) \cdot M_{soil,er} \cdot f_{om} \cdot f_{om,N} \cdot f_{er,sed} \qquad [33]$$

Nitrogen associated with eroded sediments is already integrated into the TN leaching model. The data used to calibrate that model included all soluble and particulate forms of nitrogen. Thus, although the nitrogen flux with eroded sediments can be calculated separately as described by eqn. [33], it is not included as a separate term in the implementation of this model.

N_2O Emissions

Nitrous oxide, a major contributor to greenhouse gas emissions, is released from agricultural systems during nitrification and denitrification reactions. Most studies lump together all sources of nitrous oxide to predict that ~1.25 - 1.5% of the total fertilizer nitrogen used is emitted as N_2O (e.g., Matthews, 1994, in Cary et al., 2002; Brentup et al., 2000; Wang, 1999), with additional emissions from soil mineralization process. Alternatively, it can be estimated more directly as a fraction of the total amount of nitrogen that undergoes nitrification (note – L_{ni} includes soil mineralization in this model) and denitrification reactions (deVries et al., 2003). This approach already accounts for variable denitrification rates due to soil type and moisture content.

$$L_{N2O} = f_{de,N2O} \cdot L_{de} + f_{ni,N2O} \cdot L_{ni} \qquad [34]$$

For sand/loess/clay soils, these factors are estimated by deVries et al. (2003) as $f_{ni,N2O}$ = 0.005 – 0.02 and $f_{de,N2O}$ =0.01 – 0.06. Higher values within this range of $f_{de,N2O}$ would be appropriate for peat soils.

NO_x Emissions

Nitrogen oxide (NO) is also released during nitrification and denitrification reactions. An approach similar to that described above for N_2O can be used to estimate NO emissions.

$$L_{NO} = f_{de,NO} \cdot L_{de} + f_{ni,NO} \cdot L_{ni} \quad [35]$$

deVries et al. (2003) suggests factors $f_{ni,NO}$ = 0.01 – 0.03 and $f_{de,NO}$ =0.01 – 0.02, independent of soil type or moisture content.

NO_x, predominantly NO_2, is also emitted during farming activities when diesel fuel and gasoline are combusted by tractors. These emissions are described in the section on energy inputs and use.

Nitrogen Emissions

Although nitrogen gas is not an environmental concern, quantifying this flow is required to provide closure of a mass balance. It is assumed here that all nitrogen that is denitrified, but not emitted as NO or N_2O is converted into nitrogen gas.

$$L_{N2} = (1-f_{de,N2O} - f_{de,NO}) \cdot L_{de} \quad [36]$$

Calibration of Nitrate Leaching Model

The eastern Iowa watershed system was selected as a geographic boundary due to the extensive data for TN and NO_3-N fluxes from E. IA rivers to the Mississippi. The availability of these data provides an excellent means of calibrating our nitrogen flow model against real data. Leaching from fields and in-stream denitrification processes were both considered in this calibration process.

Data sets quantifying the total nitrogen (TN) load are available from the USGS National Ambient Water Quality Assessment (NAWQA) (Becher et al., 2001) and the NOAA/USGS Hypoxia in the Gulf of Mexico study (http://co.water.usgs.gov/hypoxia/html/nutrients.html). Specific data that quantify the total annual loads of nitrate and/or TN from the three primary watersheds in eastern Iowa that discharge to the Mississippi River were used. Neither of these data sets provides both TN and NO_3-N fluxes for all three watersheds of interest, however. TN loads are presented in Table 9. These include both those directly available through the datasets as well as those estimated based on the relative contributions of the different watersheds to the total. Values for the Cedar River – which is included within the overall Iowa River watershed – are included because they provided the best correlation to the fraction of TN load associated with the Wapsipinicon River. For the three years that a complete data set for all four watersheds was available (Becher et al., 2001), it was determined that the Wapsipinicon TN loads were approximately one-third (0.35±0.06) of the Cedar River values. This fraction was used to estimate the total nitrogen load to the Mississippi River during the period 1988-1995.

Table 9. TN Loads (mt/yr) to the Mississippi River from E. Iowa Watersheds

Year	Wapsipincon	Iowa	Skunk	Cedar
1988	2,200	20,179	5,087	6,290
1989	1,227	7,616	2,189	3,508
1990	9,514	69,219	22,483	27,196
1991	19,618	96,879	22,729	56,079
1992	13,841	88,422	23,860	39,564
1993	35,784	199,911	51,287	102,290
1994	9,336	46,969	8,026	26,688
1995	8,284	54,321	15,877	23,680
1996	9,990	57,600	30,000	31,900
1997	21,700	75,100	23,500	52,200
1998	31,100	154,000	49,400	97,000

1996 – 1998 data from Becher et al. (2001)
1988 – 1995 data for Skunk, Iowa and Cedar Rivers from NOAA/USGS Hypoxia study
1988 – 1995, Wapsipinicon R. – Estimated based on 0.35 * $TN_{(Cedar)}$

As an average over the years, nitrates contribute 78, 91, and 90% of the nitrogen in the TN flux for the Iowa, Skunk and Cedar Rivers, respectively, although the percentages range from 50-100%. With this range of uncertainty, it was difficult to predict NO3-N fluxes for the Wapsipinicon for the years no NO3-N data were available (1988-1995). Thus, we could not directly calibrate potential leaching model (Table 7) to available NO3-N data, and chose to focus on TN instead.

Becher et al. (2000) provide an estimate of nitrogen sources in Eastern Iowa. They show that a combination of FN and animal manures contribute over 90% of the TN discharged to the Iowa and Skunk Rivers. Thus, we assumed here that the inclusion of animal manures as a TN source would enable us to perform the most accurate calibration. Becher et al.'s (2000) estimate for the animal manure generated in this region is used to identify potential sources of TN to the watershed system (L_{am} = 59,600, 21,800 and 12,800 mt TN/yr applied to Iowa, Skunk and Wapsipinicon watersheds, respectively in 1996). The total that the animal manures contribute (92,400 mt TN) adds approximately 34-40% additional nitrogen to the watershed in addition to the inorganic FN application. FN used in this region was calculated from published county totals for corn and soy acreage, and statewide average amounts of FN used and fraction of farms applying FN to their crops (see additional details in Table A-2). Two key assumptions were made in this analysis:

1. The same fraction of nitrogen leaches from animal manures as inorganic commercial fertilizer in a given year. This assumption is also use in the IPCC nitrogen flow model.
2. Denitrification of NO_3 in rivers also varies some with rainfall. The NO_3 fraction of the TN was required to estimate denitrification. This fraction was defined as an average for watershed discharge data available 1988 – 1998, with a weighted average between the three rivers ($f_{TN,NO3}$ = 0.82). In contrast, Goolsby et al. (2000) report that nitrate accounts for 61% of the total nitrogen discharged from the Mississippi River to the Gulf of Mexico.

The model defining the load of TN to the Mississippi River (L_{MS}) can be represented as:

$$L_{MS} = \{(L_f + L_{am}) \cdot f_{TN,SW}\} - \{(L_f + L_{am}) \cdot f_{TN,SW}\} \cdot f_{TN,NO3} \cdot f_{de,ri} \qquad [37]$$

Where the term in the {} is the TN leached within the watershed and the second term defines the NO_3-N fraction that is lost due to denitrification within the watershed. As described above, $f_{TN,SW}$ was estimated as:

$$f_{TN,SW} = \text{intercept} + \text{slope} * \text{rain(mm)} \qquad [38]$$

where the intercept and slope were determined by regression analysis of field and drainage basin scale data from the eastern Iowa region. Initially rainfall data from Cedar Rapids, which is centrally located within the region, was assumed representative of the region. Due to the very high level of sensitivity for this parameter, however, the calibration process was changed to use average rainfall values for each county to estimate the fraction of FN leached on a county-by-county basis.

Values for the calibrated model parameters used are included in Table 10. The model estimates for L_{MS} were very sensitive to the slope, intercept and rain data used in the regression equation. Very small modifications in the slope and intercept helped substantially to improve the quality of the fit. Application of the high denitrification rates ($f_{de,ri}$) suggested by deVries et al (2002) for the Netherlands resulted in very low estimates of L_{MS}. These rates were lowered for high flowrate years to account for the high TN fluxes discharging to the MS river. The model was not very sensitive to $f_{TN,NO3}$, so this parameter was not adjusted. The model was calibrated by minimizing the error in the difference between actual and modeled cumulative discharge to the Mississippi River over the period 1988-1998. The calibrated model estimates are within 1% of the actual cumulative TN load.

Table 10. Values Used to Calibrated the Leaching/Denitrification Model

Parameter	Value		Comments
	Initial	Calibrated	
Intercept	-0.457	-0.48	Highly sensitive to small changes
Slope	0.000881	0.00103	Highly sensitive to small changes
$f_{TN,NO3}$	0.82	0.82	Not sensitive
$f_{de,ri}$	0.3	0.2	For years with high flow (Table 2). Sensitive, required lower value to estimate high TN loads
$f_{de,ri}$	0.4	0.4	For years with low flow (Table 2).

Figure 13 presents the TN load to the Mississippi River estimated with the calibrated model (Table 10) as compared with data from water quality measurements. With a few exceptions, the calibrated model represents both the trends among years and the absolute value of the TN load.

For individual years, the greatest percent errors were for 1988 (-57%) and 1989 (+266%) (Figure 14). Estimates for six of the years were within 25% of the loads estimated from water quality data. The highest absolute error and the largest deviation in the overall trends in the water quality data are associated with the estimate for 1990. The model estimate for 1990 is consistent with observations from Walnut Creek (Jaynes et al., 1999) and the overall middle Mississippi River loads that suggest that the nitrate load that year was high due to a wet year following two dry years with poor yields and little or no leaching. The data presented in Figure 15 shows that 1990 was indeed an anomalous year. The total nitrogen load to the Mississippi River was much lower than the general linear trend between loads and rainfall that fits other years quite well. Thus, the linear model between nitrogen discharge and rainfall does not adequately represent all years.

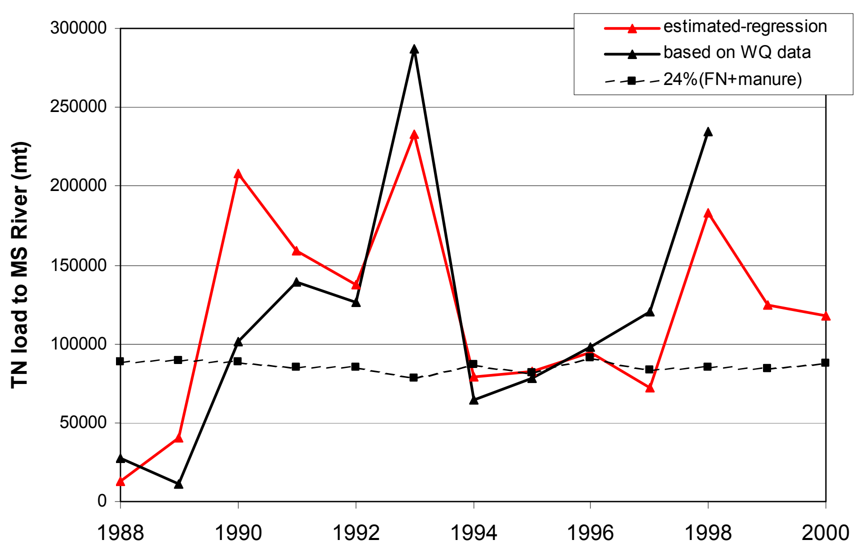


Figure 13. Actual TN flux to the Mississippi River and calibrated model including leaching of FN and animal manure and in-stream denitrification. The method used by GREET, which employs a constant fraction (24%) of FN to estimate leaching (modified here to also include animal manure), is included for comparison.

The approach used by models such as GREET (Wang 1999) (leaching = 0.24 · L_f) provides an excellent estimate for "average" years (1994-1997) (Figure 14). However, they do not capture the wide variability in leaching rates as correlated to rainfall. Errors with constant value for the fraction leached are as high as 700% overestimate (1989), although the overall cumulative error is only 13%.

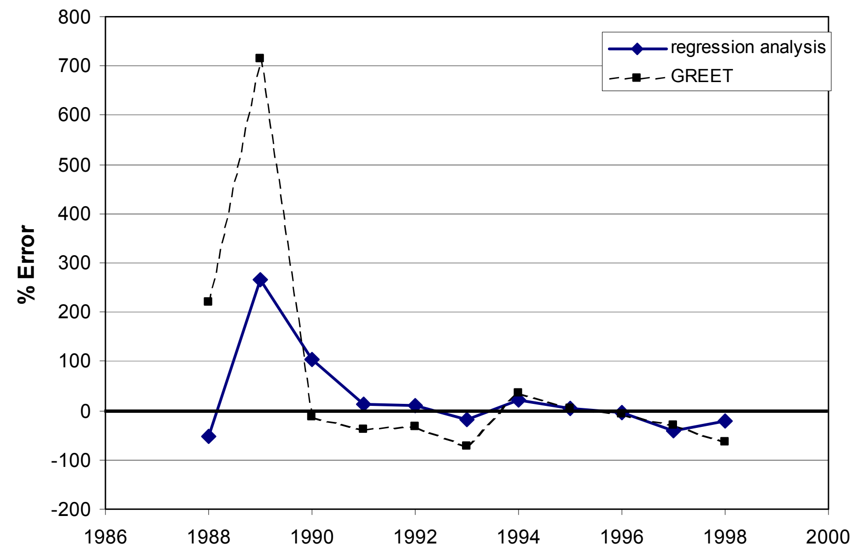


Figure 14. Percent error in rainfall-based regression analysis and GREET models for predicting TN discharged to the Mississippi River.

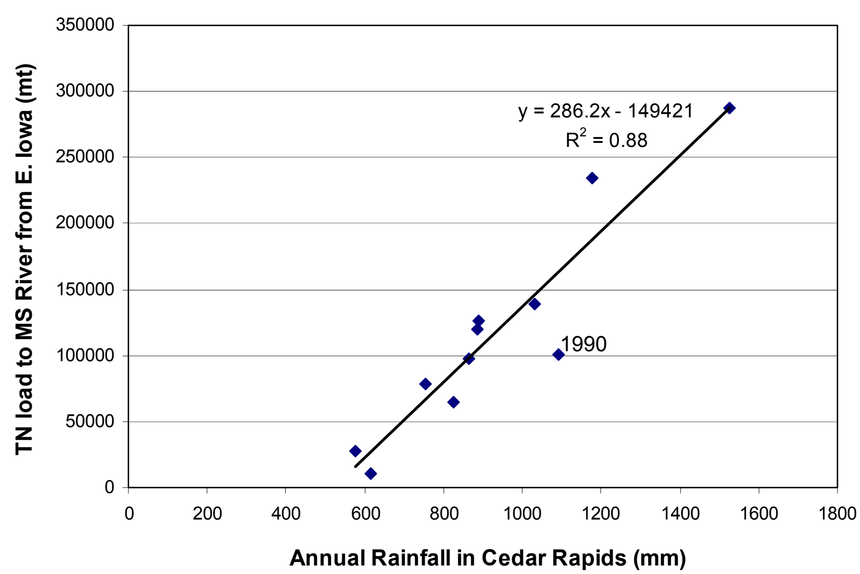


Figure 15. The actual TN load determined from water quality data (Table 9) in 1990 is significantly lower than the general linear increasing trend with rainfall.

Phosphorus Flow Model Development

The quantification of key phosphorus flows thought the C-S agricultural system is much simpler than for the nitrogen flows. It is assumed that inorganic fertilizer phosphorus (FP) is the only inflow. Outflows include P lost with the crop harvest and P leached to surface waters. Due to the strong sorption of P to soil minerals, it is assumed that no P leaches into groundwater.

Fertilizer application

The phosphorus fertilizer load is analogous to that described for nitrogen (eqn. [14]). This equation includes that applied to corn, soy and the additional fertilizer required to replace the phosphorus lost from the system with the previous year's stover harvest.

$$L_f^P = A_{c,p} \cdot N_{f,c}^P \cdot f_{f,c}^P + A_{c,H} \cdot \left(H_{st,har}^{(t-1)}\right) \cdot f_{st,P} + A_{soy,p} \cdot N_{f,soy}^P \cdot f_{f,soy}^P \qquad [39]$$

Appropriate historical values for the area planted ($A_{i,p}$) are multiplied by the application rates ($N_{f,i}$) and fraction of acres fertilized ($f_{f,i}$) to quantify the loads due to fertilization. Subscripts c, st, and soy refer to corn, stover and soybeans, respectively and the superscript P defines these as phosphorus-specific values.

Removal with Crop

Agricultural data quantifying crop yields and phosphorus content in plants can be used to define the phosphorus leaving the system with the plant matter.

$$L_{harv}^P = A_{c,H} \cdot (H_c \cdot f_{c,P} + (H_{st,harv}) \cdot f_{st,P}) + A_{soy,H} \cdot H_{soy} \cdot f_{soy,P} \qquad [40]$$

Where H indicates the mass harvested, $f_{i,P}$ is the mass fraction of phosphorus in the material harvested.

Leaching to Surface Waters

An approach similar to that described above for nitrogen leaching was used to estimate the amount of phosphorus leached as a function of rainfall and to calibrate this model to actual data for phosphorus discharged from the three E. Iowa watersheds to the Mississippi River. Flows are quantified as total phosphorus (TP), which includes P in both soluble and particulate forms.

The smaller-scale study by Klatt et al. (2003) provides some data used as a basis for the leaching model developed here. They measured P inflows to Clear Lake in NE Iowa over a two-year period. Clear Lake is within the geographic boundary considered here and is a good example of a water body adversely impacted by eutrophication. The TP loads were significantly higher during the year with a higher total rainfall (Figure 16). This suggests that model of the same form developed for nitrogen can be used to determine the TP load from this system to surface water bodies.

$$L_{SW}^P = L_f^P \cdot f_{TP,SW} \qquad [41]$$

Where, the fraction leached is defined as a linear function of rainfall:

$$f_{TP,SW} = \text{intercept}^P + \text{slope}^P * \text{rain(mm)} \qquad [42]$$

Although these data are too few to generate a reliable leaching model, they did provide a starting point to compare the fraction of TP inputs to the E. Iowa watershed system that are leached based on measured water quality data provided for the Wapsipinicon, Iowa ands Skunk Rivers. TP loads from these rivers to the Mississippi River are presented in Table 11. Values for the Wapsipinicon (1988-1995) were estimated as a fraction of the total loads from other rivers.

Additional TP inputs from animal manure were available for the Iowa and Skunk Rivers in 1997 (Goolsby et al., 1999). Extrapolating these to the entire area considered here provided an average estimate of 32,088 mt P from manure generated in this area every year.

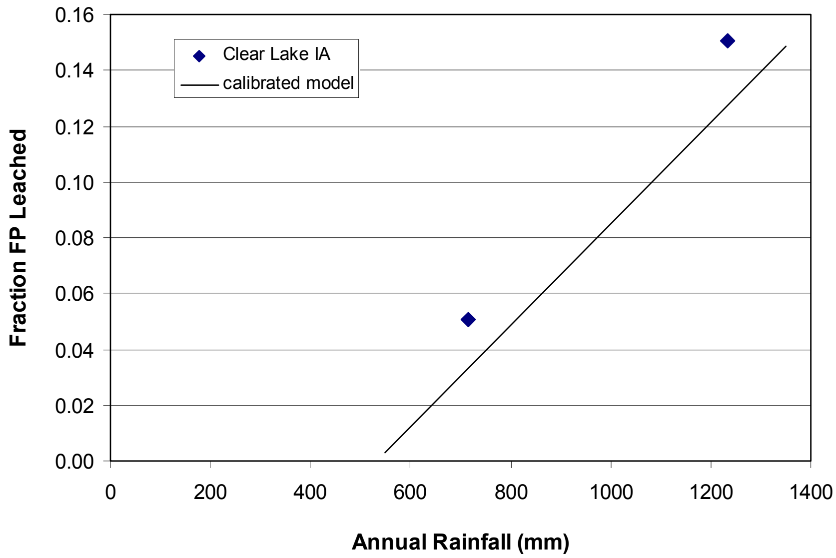


Figure 16. The fraction of FP that leaches to surface waters can be expressed as a linear function of rainfall. Limited data available from Klatt et al. (2003)

Table 11. Measured TP Loads (mt) from the E. Iowa Watersheds to the Mississippi River

Year	Wapsipinicon River	Cedar River	Iowa River	Skunk River
1988	81	236	913	146
1989	52	155	557	118
1990	354	844	3108	1711
1991	416	1778	3777	1112
1992	326	1027	2893	1295
1993	1158	3520	10341	4673
1994	172	683	1711	365
1995	248	653	2177	1140
1996	540	1560	2360	3960
1997	440	1860	3120	990
1998	476	2470	3930	4140

1996 – 1998 data from Becher et al. (2001)
1988 – 1995 data for Skunk, Iowa and Cedar Rivers from NOAA/USGS Hypoxia study
1988 – 1995, Wapsipinicon R. – Estimated based on 0.062 * $TP_{(Cedar+Skunk+Iowa)}$

The calibrated model relationship between the fraction of TP leached and rainfall is

$$f_{TP,SW} = -0.097 + 0.000182 * rain(mm) \qquad [43]$$

This compares quite well with the limited data available from Klatt et al. (2003) (Figure 16). The predicted lower fractions leached relative to Klatt et al.'s (2003) measurements could indicate losses of TP in rivers between the discharges from the field and ultimate discharge to the Mississippi River. Settling of particulate forms of P or consumption of bioavailable forms by aquatic species would both be expected. The overall fraction of the TP leached ~ 1-14% includes the range typically assumed (<1-5%; Smil 2000), although fractions predicted by eqn. [43] also includes much higher values to account for increased leaching rates during wet years.

Estimates of the TP load to the Mississippi River from the three eastern Iowa watersheds based on eqn [43] are shown in comparison with measured data on Figure 17. Trends in the estimated TP load track well with observations. The final slope and intercept presented in eqn. [43] were adjusted to minimize the errors between observed and estimated TP loads. The errors between the estimated and observed loads range from –80% (1988) to +91% (1989), although it is apparent that the absolute errors in these years was small. The cumulative 11-year error was +2.1%. The predicted TP load to the Mississippi River is extremely sensitive to the slope in this equation.

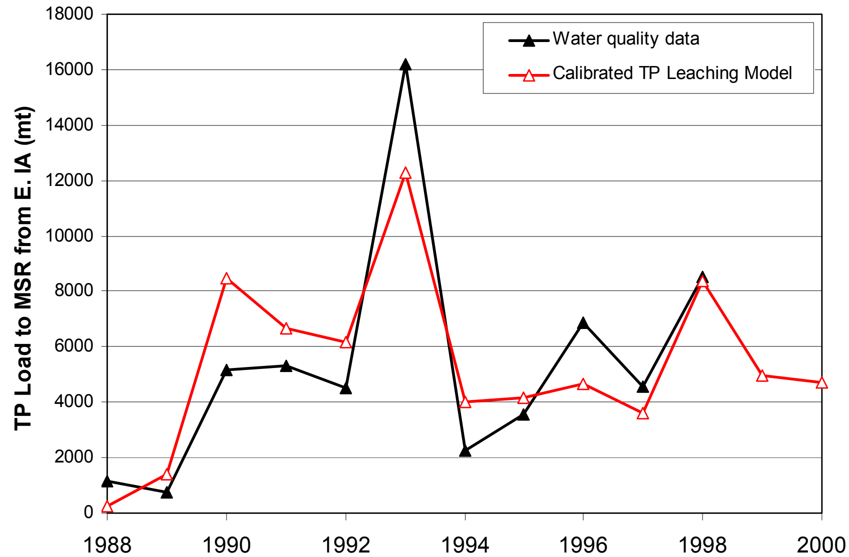


Figure 17. Actual TP flux to the Mississippi River and calibrated TP leaching model, which includes soluble and particulate P.

Potassium Model

The quantification of key potassium fertilizer flows accounts only for the potassium added as fertilizer and that removed with the grain and stover. It is assumed that the potassium does not partition to air or water environmental compartments in any appreciable quantity.

The fertilizer load equation is identical to that for nitrogen and phosphorus. It includes that applied to corn and soy, and the additional fertilizer required to replace the phosphorus lost from the system with the previous year's stover harvest.

$$L_f^K = A_{c,p} \cdot N_{f,c}^K \cdot f_{f,c}^K + A_{c,H} \cdot \left(H_{st,har}^{(t-1)}\right) \cdot f_{st,K} + A_{soy,p} \cdot N_{f,soy}^K \cdot f_{f,soy}^K \qquad [44]$$

Agricultural data quantifying crop yields and potassium content in plants can be used to define the potassium leaving the system with the plant matter.

$$L_{harv}^K = A_{c,H} \cdot (H_c \cdot f_{c,K} + (H_{st,harv}) \cdot f_{st,K}) + A_{soy,H} \cdot H_{soy} \cdot f_{soy,K} \qquad [45]$$

The terms in both these equations have been defined previously, with super and subscript K used to identify potassium specific values.

Energy Use and Emissions on the Farm

Energy used in farming activities and their associated emissions were quantified. This information was required to define the total nitrogen emissions to the atmosphere and the acidification impact. The energy flows are also used to define greenhouse gas emissions from the field so the relative benefits of GHG reductions can be compared to the detrimental effects of eutrophication and acidification.

The energy flows in the crop production stage of the life cycle (e.g., farming) only include the diesel, gasoline and motor oil used in the planting, cultivation and harvesting of corn, soybeans and stover. Fuel used for corn and soybean farm activities is included in Table 12. These data were developed specifically for current conditions in Iowa (Shapouri et al., 2002, Sheehan et al., 1998). Shapouri et al. (2002) included additional energy sources for corn production beyond those shown in Table 12. The inclusion of a very wide range of electricity use though between 1991 (12 kWh/ha) and 1996 (380 kWh/ha), however, increases the uncertainty in these data. The authors indicate that substantially more electric power was used in 1996 for drying purposes than in 1991. It is questionable what the best representative value for electricity use for corn would be and, thus, it is not included here. Electricity usage has not been included in prior LCAs that include soybean production (Sheehan et al., 1998).

Table 12. Energy Used in Farming Activities in Iowa

Crop	Fuel use (l/ha)			Reference, comments
	Diesel Fuel	Motor oil	Gasoline	
Corn	51.42	2.14[a]	28.05	Shapouri et al.(2002). Energy balance for corn ethanol. Fuel use taken as average of IA data for 1991 and 1996.
Soybeans	49.46	2.06	25.09	Sheehan et al. (1998), LCA for biodiesel

[a] motor oil not included in Shapouri et al. (2002). The same ratio of diesel to motor oil was assumed as used by Sheehan et al. (1998)

Lower fuel inputs are required for no till practices compared with conventional till. Data in Table 12 for corn and soy are based on current practices, which are predominantly conventional till. Detailed information provided by West and Marland (2002) were used to define a ratio of fuel used for conventional versus no till practices. They provide detailed 1990 national average data for diesel fuel consumed in each step - plowing, cultivating planting, fertilizer/pesticide application, and harvesting. The ratio of the total diesel fuel used for planting, fertilizer/pesticide application, and harvesting (NT) to the fuel used for these activities plus plow, disk and cultivation activities (CT) is 0.619. This ratio was used to adjust diesel fuel and motor oil used in the scenarios considered here with no till operations.

The diesel fuel and motor oil required to harvest stover is a function of the stover yield. It is more energy efficient to harvest higher stover yields. Data from Sheehan et al. (2002) was used to generate functions for fuel and motor oil used to collect the stover. Equations of the form:

$$\text{Fuel (l/mt)} = A \cdot \text{stover yield}^{B} \qquad [46]$$

adequately fit the data. Coefficients for this equation are included in Table 13. Gasoline is assumed to not be necessary for stover harvesting operations.

Standard emission factors were used to estimate atmospheric emissions associated with the fuel use on the farm (Table 14). Only those emissions that are associated with acidification and GHG impact categories were accounted for in this LCA.

Table 13. Coefficients for stover harvest fuel use equation
(data from Sheehan et al., 2002)

Applicable conditions	Regression Coefficients		
	A	B	R^2
Diesel Fuel:			
for stover collection < 2 mt/ha	13.40	-0.651	0.9887
stover collection ≥ 2 mt/ha	15.95	-0.414	0.9904
Motor Oil:			
for stover collection < 2 mt/ha	0.066	-0.666	0.9893
stover collection ≥ 2 mt/ha	0.080	-0.433	0.9882

Table 14. Emission factors for Fuel Combustion in Farming Tractors
(as summarized by Sheehan et al.,1998)

Fuel	Emission factors (g/MJ)[a]					
	CO	NO_x	SO_2	CH_4	N_2O	CO_2
Diesel[b]	0.32	0.89	0.12	0.0042	0.0019	75.5
Gasoline	1.14	0.63	0.0046	0.032	0.0019	67.7

[a] required conversion factors include density (ρ_g=0.72 g/mL; ρ_d=0.84 g/mL) and heat content (LHV) of the fuel (LHV_g=44.7 MJ/kg; LHV_d=43.5 MJ/kg)
[b] assumed high sulfur fuel used in farm tractors, which are considered off-road vehicles

Other Flows Included in the LCI

All energy inputs and emissions associated with fertilizer manufacture and the generation of electricity and the fuels themselves are also included. The materials and energy inputs and emissions from these upstream lifecycle stages were determined with data included in Ecobilian's DEAM database. Specific DEAM modules used are summarized in Table 15. Not all flows included in the DEAM accounting were considered in this LCA. Those that were considered here are listed in Table 16.

Carbon (CO_2, CH_4), nitrogen (N_2O, NO_x) and sulfur (SO_x) emissions to the atmosphere associated with energy use and upstream processes were quantified to provide a perspective of their contributions to global warming and acidification relative to emissions and impacts from fertilizer use and soil processes. Carbon dioxide and methane emissions associated with crop uptake and soil mineralization were not quantified here. They have been studied thoroughly and documented by Sheehan et al. (2002).

Data from the DEAM database was integrated into the MS Excel workbook model developed here. Several of these modules include electricity as an input. These were modified to integrate electricity related inflows and outflows as necessary. The Ecobilian TEAM software was not employed.

Table 15. DEAM Modules used to Calculate Upstream Flows and Emissions

	DEAM Module
401	Electricity (US, MAPP): Production.2
v1	Diesel Oil: Production
232I	Gasoline (unleaded): Production
232	Liquefied Petroleum Gas (US, LPG): Production.2
403S	Liquefied Petroleum Gas (US, Propane, Industrial Boiler): Combustion.2
111I	Lube Oil: Production
v1	Natural Gas: Production
v1 403S	Natural Gas: Combustion (Industrial Boiler)
241I	Potash (KCl): Production
241I	Superphosphate (Triple): Production
241	Ammonia (NH3): Production

Table 16. Pertinent LCA Flows from DEAM Modules

Flow[a]	Units
Inflows	
(r) Coal (in ground)	kg
(r) Natural Gas (in ground)	kg
(r) Oil (in ground)	kg
Water Used (total)	l
Outflows:	
(a) Ammonia (NH_3-N)	kg
(a) Carbon Dioxide (CO_2, biomass)	kg
(a) Carbon Dioxide (CO_2, fossil)	kg
(a) Carbon Monoxide (CO)	kg
(a) Hydrogen Chloride (HCl)	kg
(a) Hydrogen Fluoride (HF)	kg
(a) Methane (CH4)	kg
(a) Nitrogen Oxides (NO_x as NO_2)	kg
(a) Nitrous Oxide (N_2O)	kg
(a) Sulfur Oxides (SO_x as SO_2)	kg
(w) Acids (H+)	kg
(w) Ammonia (NH_4^+, NH_3, as N)	kg
(w) Nitrate (NO_3-N)	kg
(w) Nitrites (NO_2-N)	kg
(w) Nitrogenous Matter (unspecified, as N)	kg
(w) Phosphates (as P)	kg
(w) Phosphorus (as P)	kg
(w) Suspended Matter (unspecified)	kg
(w) Water: Chemically Polluted	l

[a] letters in parentheses identify these flows as resources (r), air emissions (a) and water emissions (w).

Model Implementation

Base Case

Equations ([13] - [43]) describing the nutrient flows and transformations, farm energy flows, and materials and energy flows associated with the manufacture of fertilizers and energy products were integrated into a set of Excel spreadsheet models. Input data included DEAM flows for upstream manufacturing, all fractions (Table A-3) describing the distribution of nutrients within the system, and site-specific data for the Eastern Iowa agricultural system (Table A-2). These data were collected for the period 1988 – 2000, primarily from the USDA National Agricultural Statistics Service (NASS).

The model was initially applied to the present system with no stover harvest. In this base case scenario, it was assumed that 100% of the farms use a C-S rotation with the actual land area dedicated to these crops and crop yields available at the county level. In reality, ~10% of the farms use C-C or C-C-S rotations (Brenner et al., 2001). Data were not sufficient to adequately integrate these variations into the model. Annual fertilizer application rates were only available as a state average and were assumed to apply across the entire system.

Values of some of the fractions identifying the distribution of nitrogen were adjusted to calibrate the model to the observation that long-term nitrogen content in row crop soils has been at a steady state. Thus, although there might be accumulation or depletion of nitrogen within the system in a given year, the cumulative inputs should balance cumulative outputs. This criterion was met by adjusting parameters (f_{imm}, f_{fix}, $N_{senes,i}$) to minimize the difference between these cumulative flows. These parameters had both a high degree of uncertainty in their initial values as well as significant impact on the total mass of nitrogen flowing into and out of the system.

Stover Collection Scenarios

The model was also applied to two systems (C-S and C-C) with stover harvest. These systems and the major assumptions employed are summarized in Table 17. Justification for these assumptions is included below.

To maximize stover harvest, it is assumed that all farmers switch to a no till practice. Tilling soil greatly increases soil erosion. Thus, practices that reduce tillage can be used to compensate for the potential erosion increases associated with stover removal. The change to a no till practice could, however, affect other variables as well. Grain yield and leaching rates could also be correlated to this switch in tilling practices. It has been observed that leaving residue on the field slows the warming of soil in the spring and can delay seed germination (Bjorneberg et al., 1996). However, in our case, much of the residue is removed, so the soil temperature and seed germination should not be adversely affected as in situations with no till operations and no residue removal. It is assumed that the yield is not impacted by this change.

It is assumed here that there is no difference in the fraction of FN leached between C-C, NT and C-S, CT systems. Insufficient data are available to really identify a true correlation between tilling practices or crop rotations and nitrogen leaching rates. In the carefully controlled study of Weed and Kanwar (1996), no consistent and statistically significant differences in leaching rates among tillage systems or crop were observed. Bakhsh et al. (2002) show differences in tillage, but these results are conflicting depending on the fertilization practices. In comparison with chisel plow, higher leaching rates were observed for NT practices when the fertilizer was applied in the fall, but lower leaching rates were observed when FN applied in the late spring at a rate defined by soil nitrogen concentrations. This is one of many instances where there is uncertainty in the model as it is used to forecast leaching for systems we do not have data available for calibration.

There are consistent and statistically significant data available that show that FP leaching rates are higher for NT practices. McIsaac et al. (1995) and Gaynor and Findlay (1995) both observed higher phosphate leaching rates from fields that had not been tilled. Factors of 1.6 to 5.4 times higher leaching rates have been observed. Differences among these values depend mostly on the type of soil. Given this weight of evidence, a factor is included here to increase the estimated rate of TP leaching for NT scenarios.

Although the fraction of fertilizer that leaches is assumed to not change among the scenarios, the total amount of FN applied is different in each system. The harvest of stover is assumed to be early enough in the fall that none the nitrogen is mineralized and returned to the soil. Thus, additional FN was added to replace this loss. This amount was adjusted to account for the additional loss of some FN immediately to volatilization as was assumed by Sheehan et al. 2004.

The switch from a C-S to a C-C rotation results in the loss of benefits associated with nitrogen fixation by the soy plants, requiring additional FN applied in a C-C system in comparison with the corn years of a C-S rotation. Gentry et al. (2001) identify two processes by which the presence of soy in the preceding year benefits corn: 1) there is greater mineralization of soil nitrogen in the year following soy; and, 2) there is a greater quantity of residual inorganic nitrogen in the soil associated from residue of soy nodules created during nitrogen fixation. These differences were accounted for in the model by adjusting the rate of soil mineralization (eqn. [16]) and adding additional FN. The general guideline suggested by Kurtz et al. (1984) was used here – FN requirements for corn following soy are 45 kg/ha less than for corn following corn.

Soil mineralization rates are also impacted by the tilling practices. Organic carbon in soil is mineralized much more quickly when it is actively tilled, which exposes more of the soil to oxygen required for mineralization. Although this has not been documented sufficiently for nitrogen release during mineralization, it has been studied extensively for CO_2 release from soil during the mineralization process. Brenner et al. (2001) estimated the carbon accumulated due to agricultural activities in Iowa as a function of several different variables. Comparing their estimate of the accumulation of carbon in the soil in 1998 between conventional till practices (1.57×10^6 mt C) and no till practices (2.42×10^6 mt C), there is approximately 46% less mineralization associated with no till practices in comparison with conventional till. This percentage was used to reduce the fraction of SOM mineralized each year (f_{min}, eqn. [16]).

Allocating Flows Among Crops

In an LCA, it is important to not only quantify flows through the environmental compartments, but also to allocate specific fractions of the total flows among the various products and by-products generated. This is particularly difficult in an agricultural system when there are synergistic affects associated with crop rotation. Van Zeijts et al. (1999) address this issue for allocating fertilizer among various crops in the Netherlands. Their overall approach is used here as well. They assume:

- All nitrogen can be allocated to the crop for which it is applied in the year that it is applied
- All P and K fertilizers should be allocated among the crops over the entire crop rotation in proportion to the fraction of nutrient that is harvested with each of the grains.

They justify the differences in these approaches based on the intent of the fertilizer application and the environmental fate. Nitrogen, for example, is typically applied separately for each crop in proportion to that crops needs. Excess fertilization is avoided due to the recognition that this excess will likely be lost to leaching. Phosphate and potassium, however, behave differently. Excess nutrients applied as fertilizer for one crop conserved in the soil over a longer period. They are often applied only to one crop with the intent that they be consumed by the plants over the entire rotation. The consistency in the percentage of fields onto which FN, FP, and FK are applied (Figure 2) illustrates that indeed, many farmers chose not to apply FP and FK to fields in a soy rotation year, even though the soy beans need these for growth. These farmers rely on FP and FK remaining in the soil from the previous year.

Table 17. Summary of Systems Considered and Major Assumptions

System	Major Assumptions
Base Case – C-S rotation, conventional tillage and no stover collection	All cropland is in a C-S rotation, conventional till
	Nitrogen flows adequately described by model equation ([13] – [36]) and values of input variables (Table A-1 – A-4)
	Phosphorus flows adequately described by model equation ([39] – [43]) and values of input variables (Table A-1 – A-4)
	Erosion for all years described by RUSLE equation that uses average precipitation across years but integrates variability due to crop cover and tillage practices.
	Nitrogen flows associated with cornfields and those associated with soy fields in an individual year are allocated to these crops, respectively.
	Total system phosphorus and potassium flows are allocated between corn and soy based on their respective proportion of the total uptake and removal with the grain.
Maximum possible stover collection – C-S rotation, no till	All cropland is in a C-S rotation, no till
	Nelson (2004) analysis of average minimum required stover can be applied to estimate the allowable stover collection
	Additional FN, FP, and FK are required to replace that removed with stover the previous year. The FN addition must be adjusted for immediate volatilization
	The fraction of FN that leaches to SW and GW does not change due to the stover harvest or tillage practices
	Phosphorus leaching is higher with no till practice than CT
	Nitrogen, phosphorus and erosion flows associated with the stover harvest can be defined as the difference between flows in this scenario and the base case.
	Less diesel fuel and motor oil required for NT operations
Maximum possible stover collection – C-C rotation, no till	All cropland planted in corn plus soybeans is now planted in continuous corn, no till
	Nelson (2004) analysis of average minimum required stover can be applied to estimate the allowable stover collection
	FP and FK application rates per year are less than for C in a C-C rotation than in a C-S rotation since some of the fertilizer applied to corn was allocated to soy. FP and FK rates determined by allocation procedure in the C-S rotation are used for C-C system.
	Additional FN, FP and FK are required, replacing that removed with stover the previous year. The FN addition must be adjusted for account for that lost immediately by volatilization
	Additional FN (45 kg/ha) required due to the loss of the benefits of rotation with soy (Kurtz et al., 1984)
	The fraction of FN that leaches to SW and GW does not change due to the stover harvest or tillage practices
	Phosphorus leaching is higher with no till practice than CT
	Overall mineralization rates are less in a C-C vs. C-S system (Eqn. [16]) (Gentry et al., 2001, West and Marland, 2002)
	Nitrogen, phosphorus and erosion flows associated with the stover harvest can be defined as the difference between flows in this scenario and the base case.
	Less diesel fuel and motor oil required for NT operations

The suitability of the Van Zeijts et al. (1999) approach for nitrogen allocation between corn and soybeans could be argued. Nitrogen fixation by the soy increases the nitrogen content in the soil and the soil mineralization rate during the corn years, thus providing extra nitrogen that would not otherwise be available (Gentry et al., 2001). Likewise, excess nitrogen applied to the soil during a corn year and immobilized with in soil organic matter could become available to soy plants in the following year (Baker and Timmons, 1994). The "carry over" of nitrogen fertilizer is indeed observed in leaching rates that are generally no different on a kg/ha basis in years that corn is grown versus years that soy is grown (Weed and Kanwar, 1996; Bakhsh et al., 2002). Thus, the approach used by van Zeijts et al. (1999) for nitrogen allocation is simplistic.

An alternative approach for allocating the amount of FN leached from corn and soy fields could be based on field data that quantifies actual leaching rates from cornfields versus soy fields. Data for leaching rates from the Nashua IA agricultural testing facility were used to determine a suitable allocation of nitrates leaching from corn fields and soybean fields (Weed and Kanwar, 1996; Bjornberg et al., 1996; Bakhsh et al., 2002). Data from each of these sources indicate that even though little to no FN is applied to soy fields, nitrate leaching rates (kg/ha) in a given year are approximately equal from each type of field. Statistical analysis showed that there are no trends in the fraction of nitrate leaching from corn as a function of year or tillage practices (α=0.05). Thus, all data were lumped together (n=42) to show that $f_{N\text{-}leach,c}$=51.1% (standard deviation = ±7.18) of the total amount of nitrate that leached on a per hectare basis could be allocated to the corn. Incorporating differences in the land planted in corn and soy, the total mass of nitrate nitrogen leached to the surface water ($L_{sw,c}^{N}$; mt) can be determined as:

$$L_{sw,c}^{N} = \frac{A_{c,p}}{A_{c,p} - \left(\frac{1 - f_{N-leach,c}}{f_{N-leach,c}}\right) \cdot A_{soy,p}} \cdot L_{sw}^{N} \qquad [47]$$

where L_{sw}^{N} is the total nitrate leached to surface water (mt, eqn, [25]) and $A_{i,p}$ are the areas (ha) planted in i=soy and corn. With the larger area planted in corn each year, this results in approximately 60% of the total nitrate leached from the C-S system allocated to the corn. With the method proposed by van Zeijts et al. (1999), over 98% of the total nitrate leached is allocated to corn. This difference in allocation method also impacts the allocation of NO and N_2O between corn and soybeans due to the subsequent denitrification of nitrates in surface water.

The excess corn stover available for harvest is treated here as a waste material. Thus, only those flows that are directly attributable to the stover harvest are allocated to the stover. This includes the extra nutrients required to replace the amount removed with the stover and tractor activity associated with the stover harvest.

4. Results of Quantitative Lifecycle Inventory

Nutrient Flow Model Results

Base Case: C-S System with No Stover Harvest

The estimated nitrogen inputs and outputs from the C-S system with no stover harvest are shown in Figure 18. These plots clearly show the variability in the magnitude of these flows among the years. For example, 1988 was a very dry year with low grain yields and very small amounts of nitrogen leached. The year 1993 was recorded as a record high rainfall. Yields were low this year due to too much soil moisture and the amount of nitrate leached to surface waters was very high. This year was also the only year that sufficient excess nitrogen was available for denitrification and significant groundwater leaching. The fact that denitrification occurred is consistent with the known higher rates of denitrification in wet years, when anaerobic conditions in the soil are more likely to occur.

The nitrogen inputs and outputs balance within 8% for most of the years studied. The extreme dry year, 1988 is one exception. The difference between inflows and outflows this year (~200,000 mt; ~28% of the inputs), suggests that nitrogen was stored in the soil after this crop year. Overall, the 12-year cumulative inflows were within 1% of the outflows, illustrating that this model, with the set of input variables included in Table A-3 meets the constraint that we have a long-term steady state for the nitrogen flows.

The model used here for nitrogen flows attempts to be more mechanistic than many other models that use “emission factors” to define nitrogen flows as a direct fraction of the applied fertilizer nitrogen. These factors, however, provide a basis for comparing the results obtained here to other studies. Table 18 presents emission factors calculated from the nitrogen flow results presented in Figure 18. Values used in other studies are also included for comparison. The N_2O results are quite similar, but all other flows calculated by this model tend to be higher than the factors used by other researchers. We would expect this, especially for our (a) NH_3 LCI flow, which includes direct FN volatilization as well as plant senescencing, whereas the IPCC value only attributes (a) NH_3 flows to direct volatilization. The (a) NO emission factor could be reduced to be closer to the value used by GREET by reducing the value of $f_{ni,NO}$. This fraction is set at 0.01 in the calibrated model, which is at the low end of the range (0.01 – 0.03) suggested by deVries et al. (2003). Insufficient data are available to assess the accuracy of the value used in GREET and that calculated here. Thus, $f_{ni,NO}$= 0.01 is used regardless of the differences with other studies.

Of most interest here, the fraction of the FN that leaches from the fields and eventually to the Mississippi River appears to be high. On average, our model would estimate the load of TN to surface waters through tile and base flow to be 28-60% higher than estimates using the IPCC or GREET emission factors for leaching. However, when the amount of nitrogen leached is normalized to the area of cropland instead of the amount of FN applied, the estimated amount of TN leached from our model is quite comparable to other studies from the Midwestern United States (Table 19). Vanni et al. (2001) and Bakhsh et al. (2002) directly relate the variation in the values they measure to variability in annual rainfall, as was done in our study, and show similar ranges in the fraction of FN that is leached between dry and wet years. Given the consistency of the yields predicted with the nitrogen model used here and those measured by others, and the accuracy of the calibrated TN leaching model (Figure 13), the estimated discharges of TN from C-S fields and ultimately to the Mississippi River and Gulf of Mexico are considered realistic.

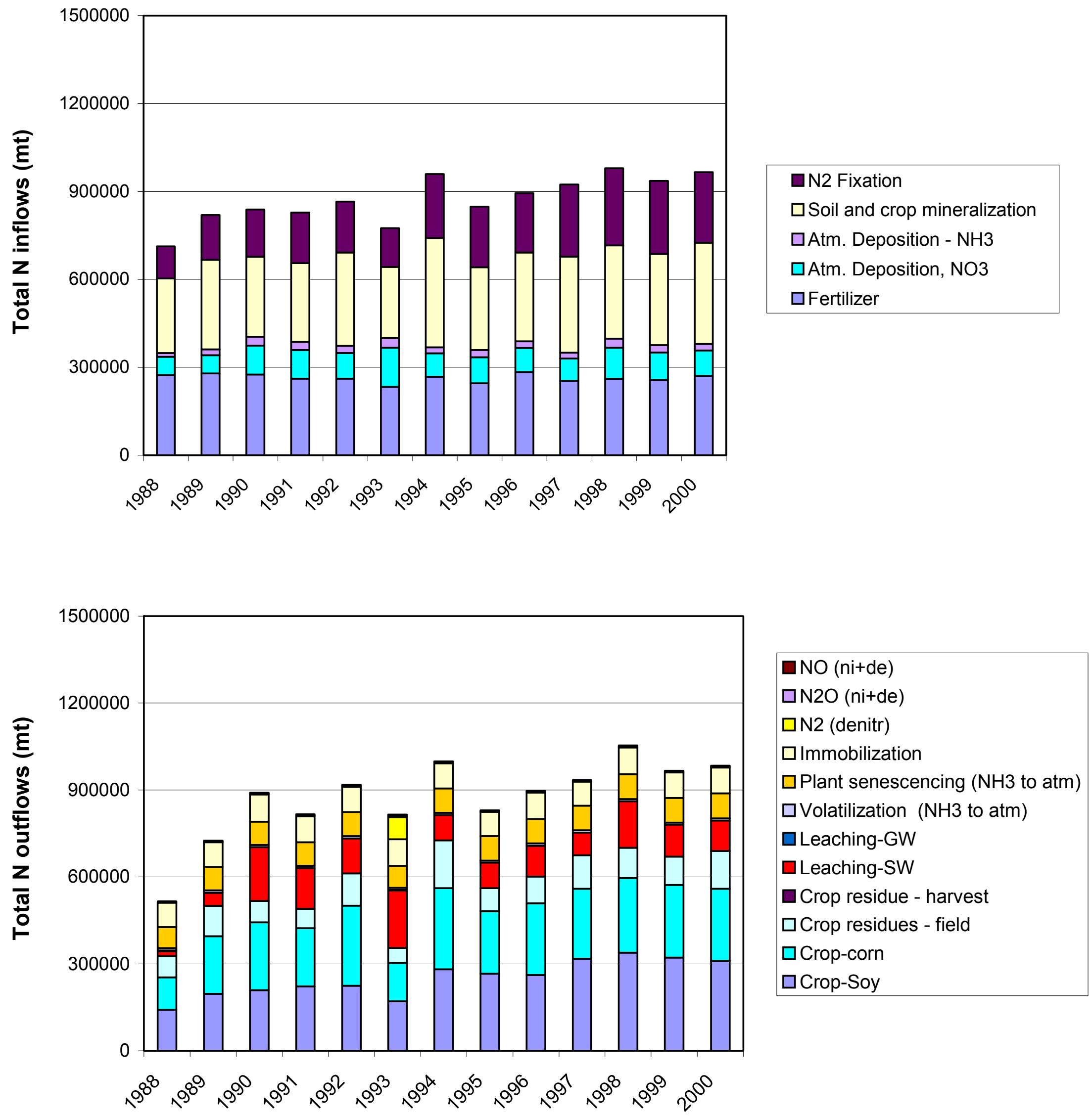


Figure 18. Nitrogen flows (mt/yr) into and out of the E. Iowa C-S system.

The distribution of nitrogen outflows between corn and soybeans is illustrated in Figure 19. This plot shows the fraction of the total C-S system flows that should be allocated to corn. Two methods were considered for this allocation: method 1 assumes all FN applied to a cornfield should be allocated to corn; and, method 2 assumes that approximately the same amount of nitrogen is leached from cornfields as soy fields (eqn. [47]). There is clearly a very large discrepancy between these two methods for several of the emissions. Thus, there is significant uncertainty in the best method to use to allocate flows for specific crops in a system with a symbiotic relationship between two crops. The most probable accurate allocation lies somewhere between these two extremes. Additional field research that provides a more mechanistic understanding of the symbiosis between crops in the C-S system is required to more accurately allocate nitrogen flows between these crops.

Table 18. Comparison of Emission Factors Determined from This Study and Those Used by Others

LCI flow	Emission Factor - Fraction of FN			Reference
	This Study		typical value	
	median	range		
(a) NH_3	0.346	0.295 - 0.373	0.100	IPCC - fraction of FN volatilized
(a) NO	0.019	0.010 - 0.030	0.008	GREET (Wang, 1999)
(a) N_2O	0.013	0.005 - 0.026	0.015 (0.01 - 0.02)	GREET (Wang, 1999)
(w) TN - SW	0.385	0.061 - 0.850	0.24	GREET (Wang, 1999)
			0.30	IPCC (1996)
(w) NO_3-N - GW	0.000	0.000 - 0.010		
(w) TN to MSR/Gu	0.308	0.037 - 0.680	0.250	Howarth et al., 1996
			0.160	Cary et al., 2002

Table 19. Comparison of Nitrogen Yields Determined from This Study and Those Measured by Others

Range of Values (kg-N/ha cropland/yr)	Comments, references
5.8 - 64	**This study (median: 32 kg/ha)**
38 - 64	NO_3-N in tile drain water, IL 30-ha field (David et al., 1997)
11 - 107	NO_3-N in tile waters, 4 different tillage systems in IA (Bjorneberg et al., 1996)
4 - 46	NO_3-N in tile waters, field study in IA (Bakhsh et al., 2002)
2 - 60	NO_3-N in tile waters, watershed study, IA, 1992-2000 (Tomer et al., 2003)
10 - 80	TN from OH watershed (Vanni et al., 2001)
10 -70	NO_3-N from OH watershed (Vanni et al., 2001)

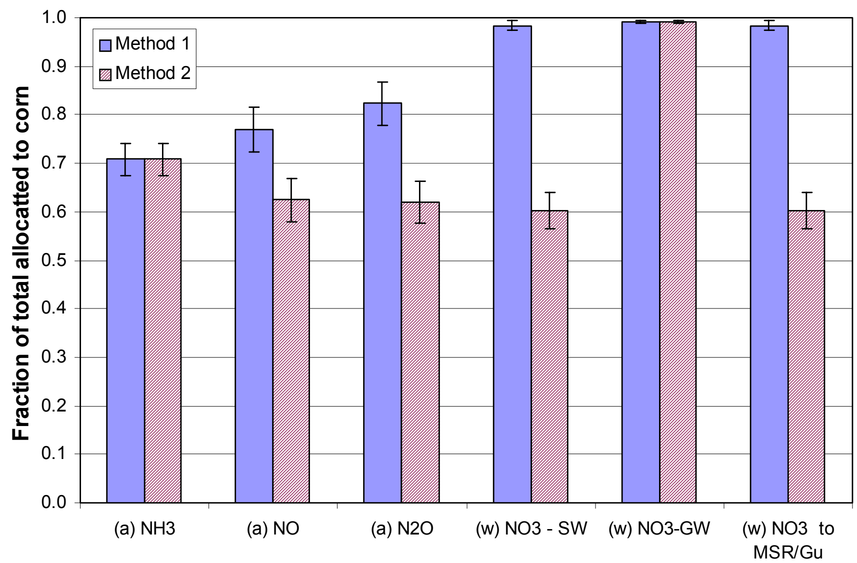


Figure 19. Fractions of nitrogen flows for C-S system that are allocated to corn. Fractions represent averages over 13-year study period with error bars indicating one standard deviation.

Phosphorus flows into and out of the C-S system in E. IA are shown in Figure 20. The phosphorus flow model is much simpler than the nitrogen model allowing the flows and allocations among the crops to be shown in Figure 20. In this case, the allocation between crops is based on the fraction of the total phosphorus removed with the harvested grain. Thus, although almost all of the FP is applied to cornfields, the distribution of FP use is more equally distributed between the crops than the nitrogen flows.

Two points should be made about this figure. First, due to the low solubility of phosphates, the amount of phosphate that is leached is very small compared with the phosphorus removed with grain. Second, there is currently a very troubling disparity between the total phosphorus inflows and outflows. Sharpley et al. (2003) and most other researchers studying phosphorus in agricultural systems indicate that FP inflows are greater than P outflows, leading to an increasing amount of P accumulated in soils. The data shown in Figure 20 suggest the opposite. The cumulative 13-year outflows in this case are almost 50% higher than the inflows. Further efforts to verify the P content in grain and FP application rates are required.

C-S and C-C systems with stover harvest

Harvestable Stover

The model was adapted to include stover harvest at the maximum possible rate to maintain acceptable levels of soil erosion. In reality, it is unlikely that the methods used to collect stover will be sufficiently efficient to harvest stover at this rate. Practical levels of stover harvest are expected to be below the rates discussed here.

Minimum required stover residue rates for the C-C system were taken from Sheehan et al. (2002). These residue requirements allowed stover harvest in all counties, all years, except 3 counties (Iowa, Keokuk and Washington) in 1988, due to the very low corn yield. The total amounts of stover harvested by year in the E. Iowa system are presented in Figure 21. The average harvestable stover yield for the C-C system was 5.66 mt/ha corn, with a range of 2.26 – 7.27 (Figure 22). This amount is less than that estimated by Sheehan et al. (2002) (6.18 mt/ha corn). They used average crop yields for 1995 – 1997, which are higher than the longer-term average, thus leading to a harvestable stover collection rate greater than determined by the analysis here, which extends over a longer period of time and includes some extreme weather years.

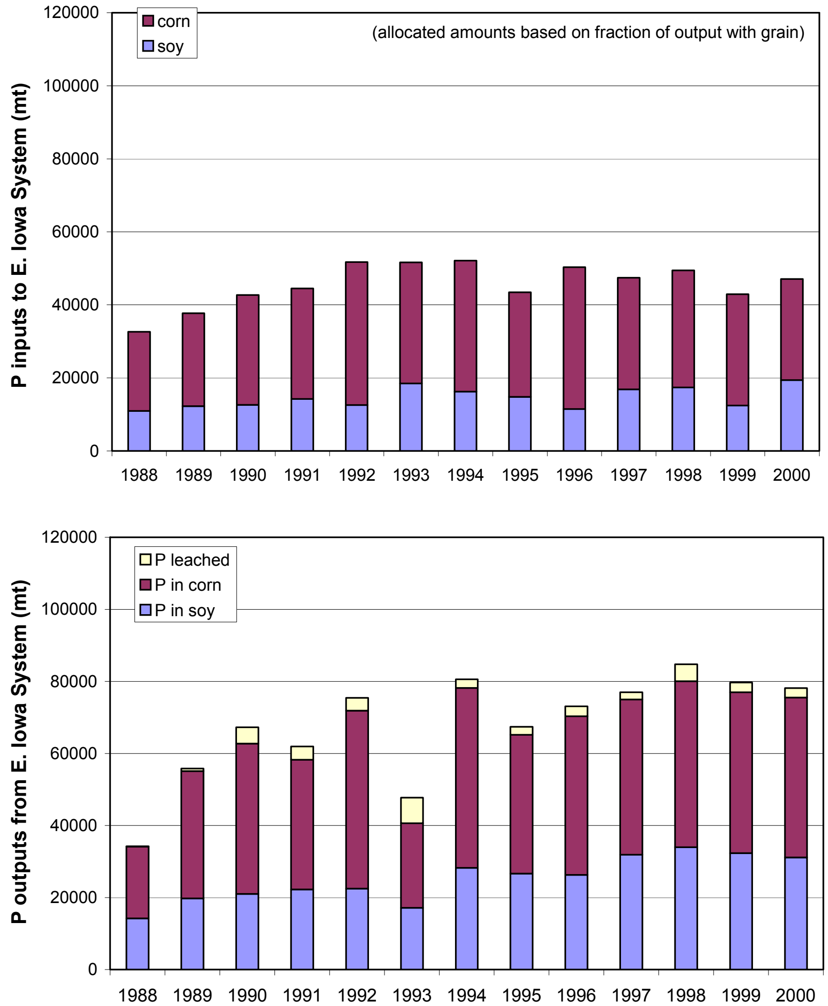


Figure 20. Phosphorus flows (mt/yr) into and out of the E. Iowa C-S system.

For C-S, the Nelson (2004) model to estimate maximum allowable removal rates was applied to each year, using the two-year average residue between the corn (t=t) and soy (t=t-1). Due to this averaging, often between years with low and high crop yields, there was sufficient reside left to allow stover harvest in most counties, all years. The stover harvest yields for the E. Iowa C-S system are included in Figure 22. The average yield for this system was 5.20 mt stover/(ha of corn harvested), with a range among the years of 2.07 – 6.74 mt/ha of corn. Stover yields for the C-C system are, on average, 10% higher than for the C-S system.

The distribution of number of counties with harvestable stover is shown in Figure 23. Stover could not be harvested in Jefferson Co. during 4 of the 13 years included in this study. No stover could be harvested from Iowa or Powesheik Counties 3 of the years, and Henry 2 of the years. Additional counties did not have sufficient residue to allow stover harvest in the extreme dry year of 1988. Of the counties with limited stover harvest, only Jasper and Jones are major contributors to the overall corn harvest and, therefore, stover harvest in this E. Iowa system (Figure 24). Limited stover harvest in other counties does not affect the overall yields substantially, although the proximity of these counties to each other in the S.E. part of the state could affect the optimal location of stover collection facilities.

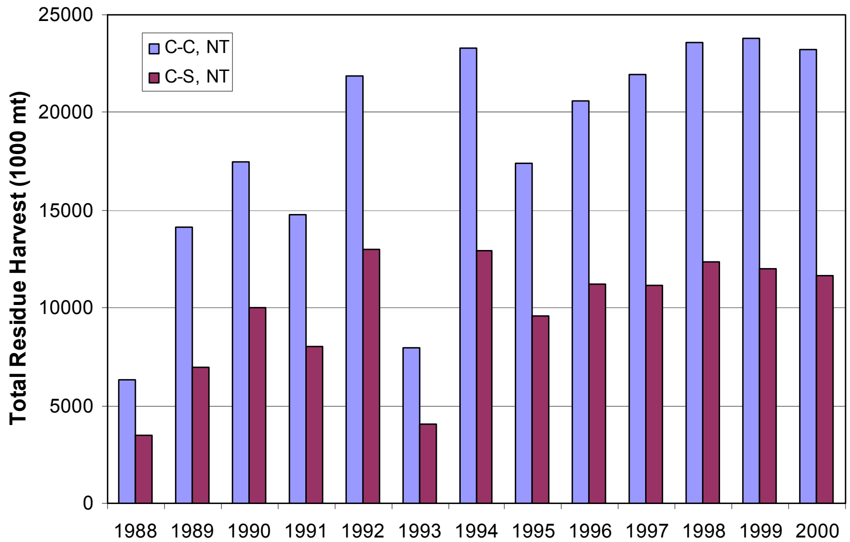


Figure 21. Total mass of stover harvested from C-C and C-S systems

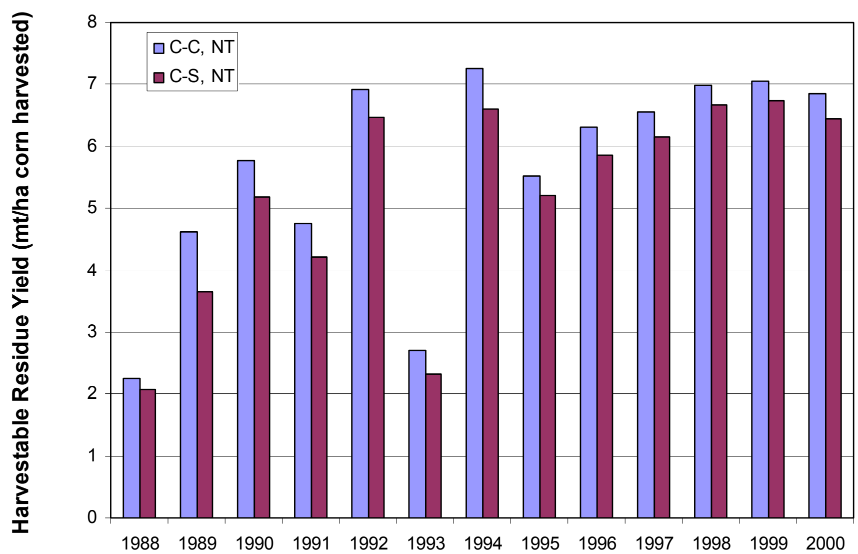


Figure 22. Stover harvest yields for C-C and C-S systems

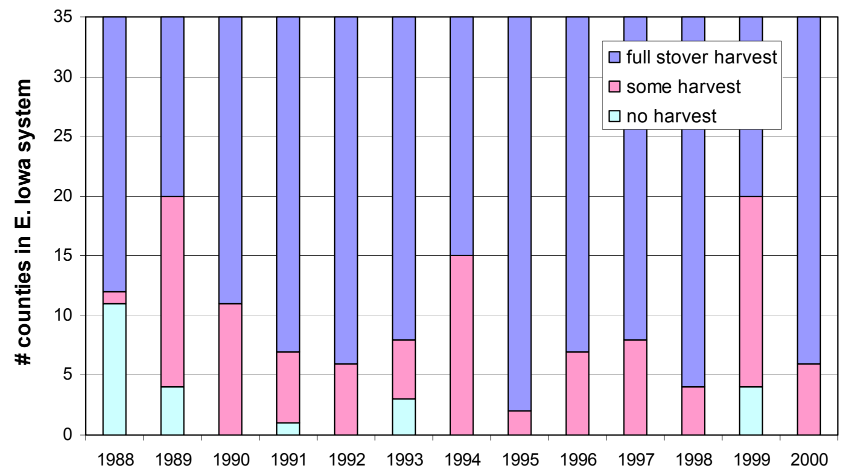


Figure 23. Distribution of the number of counties with harvestable stover.

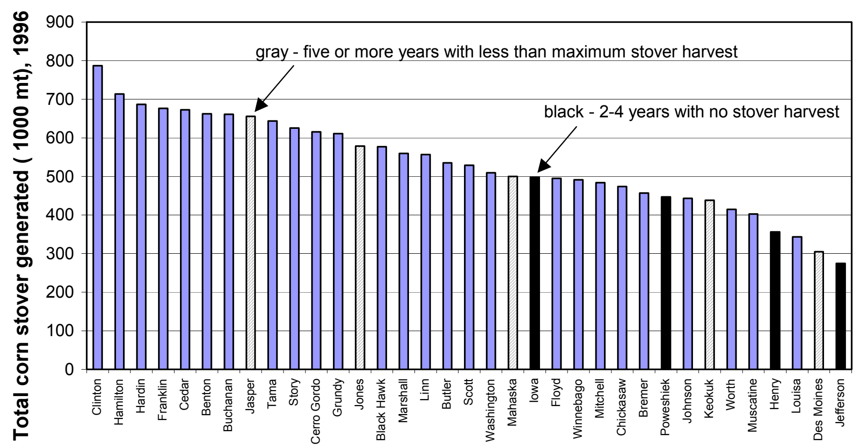


Figure 24. Distribution of corn stover production in the E. Iowa system. Black and gray striped bars indicate those counties that cannot reliably harvest stover at the maximum yield due to erosion constraints.

Nitrogen Flows

The nitrogen flow results for a C-S and C-C rotation with stover harvest are included in Figures 25 and 26. Overall, these results look similar to the system without stover harvest (Figure 18). However, there are some important differences.

Over the 13 years studied, there is a net loss of nitrogen in both of these the systems due to the stover harvest; it appears that the outputs are 17% higher for C-S and 25% higher for C-C than the inputs. Extra fertilizer was added to replace the nitrogen content in the stover that is no longer mineralized in the field and recycled. This amount of extra FN was adjusted to account for that lost by volatilization, as was done by Sheehan et al., (2002), but not lost by other sources. The amount leached into surface water is also directly correlated to the rate of FN applied. This extra nitrogen sink can account for some of the overall loss in this system. This suggests that we should modify equation [14] to increase the amount of extra FN added to replace that removed with the stover harvest to account for TN losses from the soil system due to leaching.

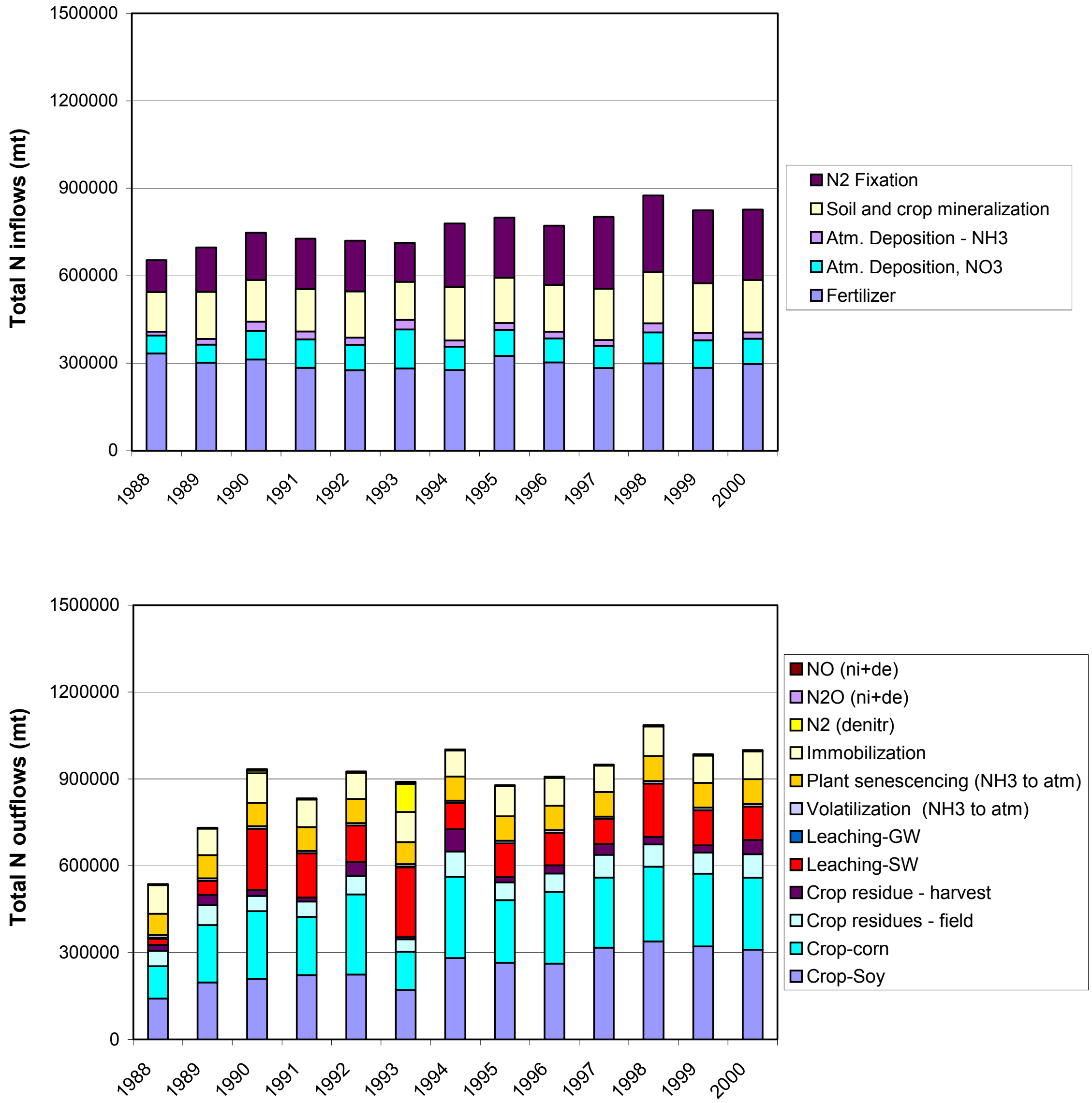
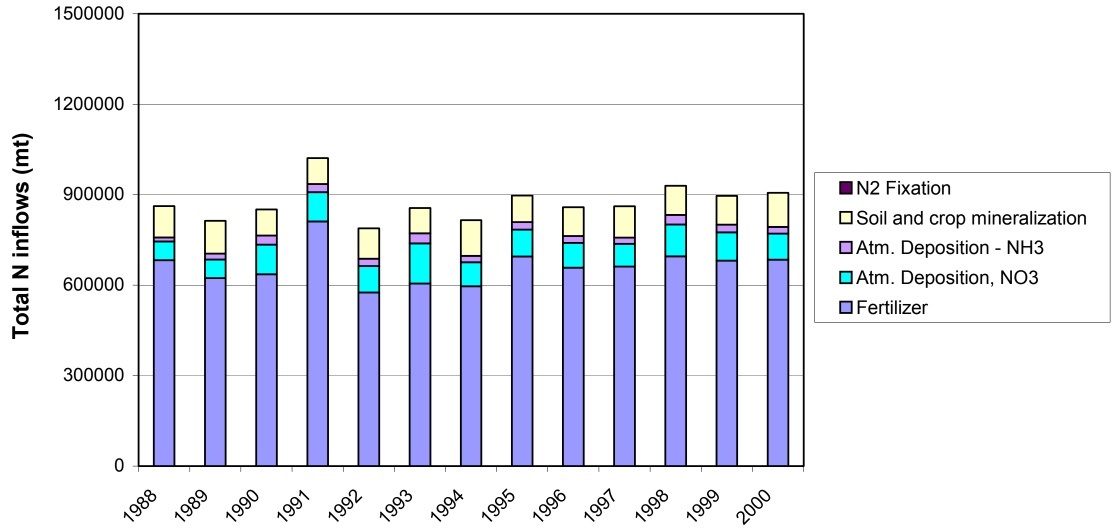


Figure 25. Overall nitrogen flows through the C-S system with stover harvest

A more significant factor – and an artifact of the way the "system" was defined here – is the reduced amount of soil mineralized in the no till scenarios. Essentially, some of the nitrogen is "lost" to the soil through immobilization, which is considered outside of the system defined here. In the base case scenario, most of the nitrogen immobilized is balanced over the years with nitrogen released through mineralization. The reduction in soil mineralization reduces nitrate inputs available to the plants, although it is still a component of agricultural soil.



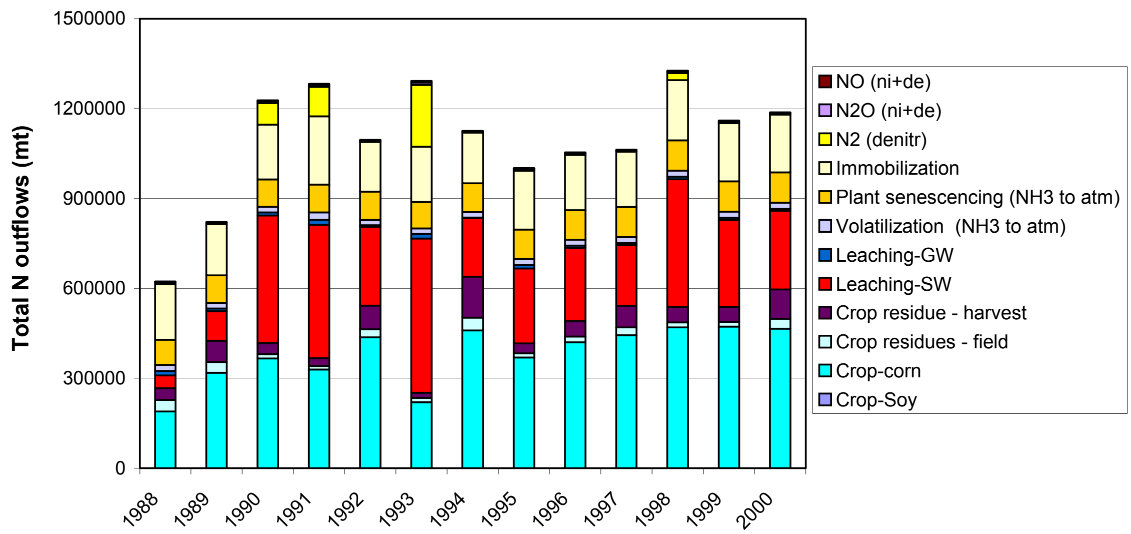


Figure 26. Overall nitrogen flows through the C-C system with stover harvest

Due to the need for an increase in the FN application when stover is harvested, there is a corresponding increase in the mass of nitrates that leach to surface water and, eventually the Gulf of Mexico (Figure 27). The very substantial increase for the C-C system is in part because the land area on which corn is grown is much higher under this scenario, thus, nearly twice as much area is treated with high FN rates associated with corn rather than soy. On average, this incremental addition above the base case system (C-S, no stover harvest) is +13% N leached to SW for the C-S-stover system (range – 3.6 – 32%) and +150% for the C-C-stover system (range - 116-215%). The incremental increase varies among the years depending on rainfall rate and the amount of stover harvested the previous year. In general, the increased nitrate leaching associated with the stover harvest is strongly correlated to years with high rainfall.

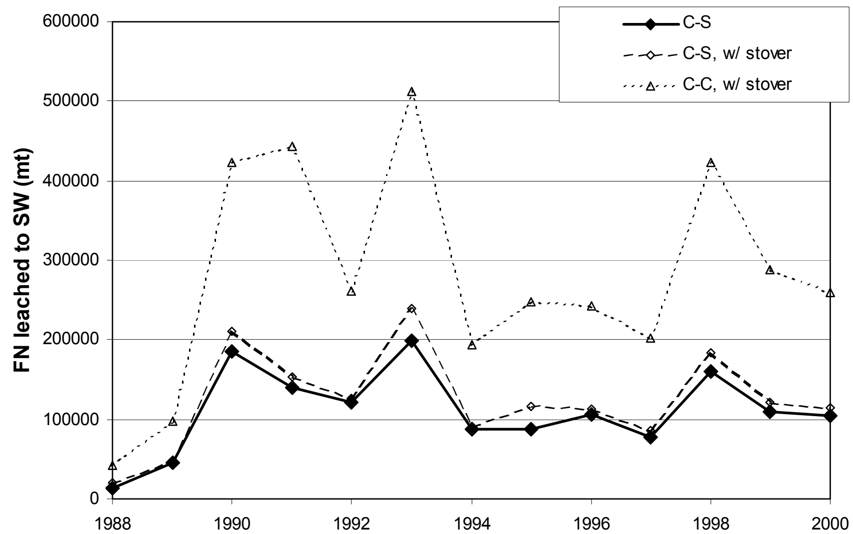


Figure 27. Nitrogen leached and discharged to E. Iowa surface waters in the three scenarios.

Normalizing these data to show the incremental additional amount of nitrogen leached per mass of stover harvested indicates that switching to a C-C system still has a more significant adverse impact in terms of nitrate pollution compared with a C-S system with stover harvest (Figure 28). The amount of nitrogen leached in the C-C system is highly dependent on the assumed requirement of extra FN due to the switch from a C-S rotation to a C-C rotation. Gentry et al. (2001) suggests that 45 kg/ha extra FN is required for corn if it is not preceded by soy the previous year. The soy provides a benefit both in terms of the soil mineralization rate and presence of soy rhizomes that release nitrogen during the corn rotation. Even without this extra FN (solid triangles in Figure 28), the amount of nitrogen leached to the surface water is substantially higher than for the C-S system with stover harvest, especially in wet years when a greater fraction of the FN is leached.

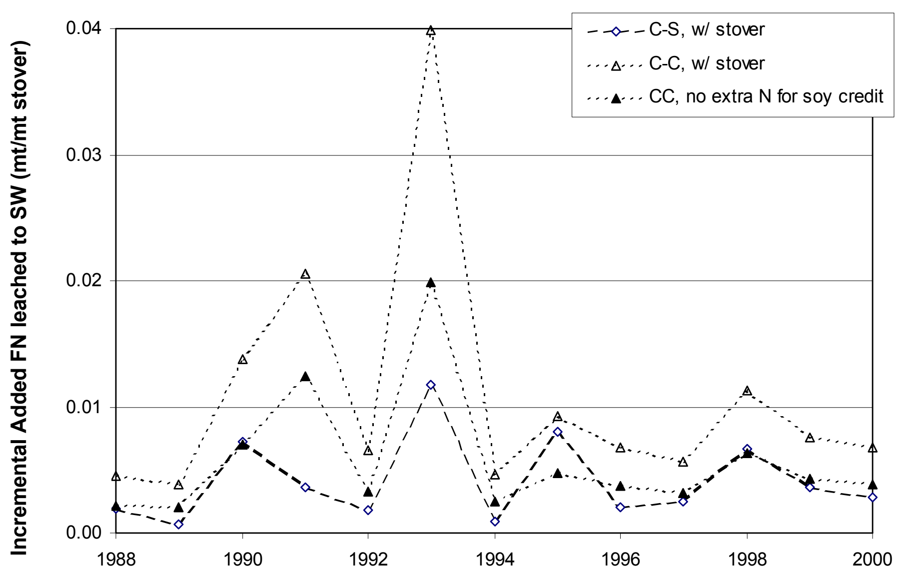


Figure 28. Incremental additional nitrogen leached to the surface water – above base case scenario (C-S) - due to the collection of stover in C-C and C-S systems

The amount of nitrogen leached to ground water (Figure 29) is very low compared to that discharged to surface waters. In fact, due to the constraints of the model used here – leaching to ground water occurs only when there is excess nitrogen in the system after crop uptake – results in a prediction that nitrate leaches to the groundwater in the base case scenario (C-S, no stover) only in those years with very low crop yields (1988, 1993).

In the C-S scenario with stover harvest, leaching to groundwater occurred only in the same two low yield years, but at a higher rate (54-69%) than the base case scenario (Figures 29 and 30). In the C-C system, the large increases in FN use resulted in sufficient nitrate available for leaching to GW in all years (Figure 29). As described above for leaching to surface water, the magnitude of the increase for the C-C system over the base case scenario is highly dependent of the amount of extra FN assumed necessary due to the lack of beneficial effects of soy (Figure 30; solid triangles – no extra FN, open triangles, 45kg/ha extra N required).

Unlike the leaching model for surface water that was calibrated with water quality data, no such calibration was possible with the estimates of leaching to groundwater. A factor of 3% of the available nitrogen was assumed (fertilizer + atmospheric – crop uptake) (eqn. [26]). This factor was based on the work of Howarth et al. (1996) who reviewed nitrogen balances on a global scale. The low value used here is reasonable considering the influence of tile drains and low permeability of the soil in eastern Iowa. In contrast, Puckett et al. (1999) assumed that all leached nitrogen infiltrated to the ground water at a more highly permeable site in Minnesota and none was discharged to the surface water (at least not directly). Overall, there is considerable uncertainty in this factor for this particular region and variability expected as a function of geological and hydrological conditions.

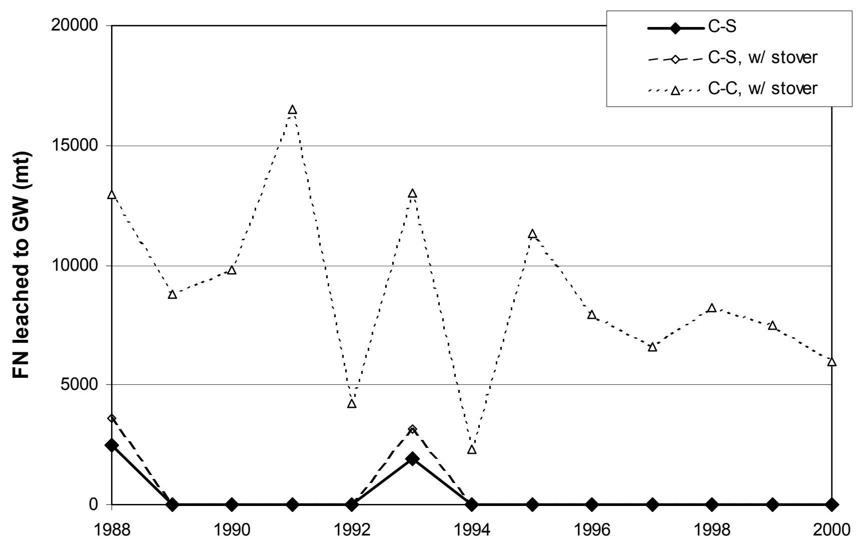


Figure 29. Nitrogen leached to ground water in each of the scenarios

Other nitrogen flows also change when the system is expanded to include stover harvest (Table 20). The percentage increase is small for most flows in the C-S with stover collection scenario; they are much more substantial for the C–C system. These values are not normalized to account for differences in the amount of stover collected between these two scenarios (C-C has nearly 2x the total stover harvest of C-S). However, it is clear that in scenarios with stover collection, emissions from the C-C scenario are still substantially higher than the C-S scenario. Results for the C-C scenario are highly dependent on the assumed extra FN required to account for the loss of the soy credit, with the worst-case value used here.

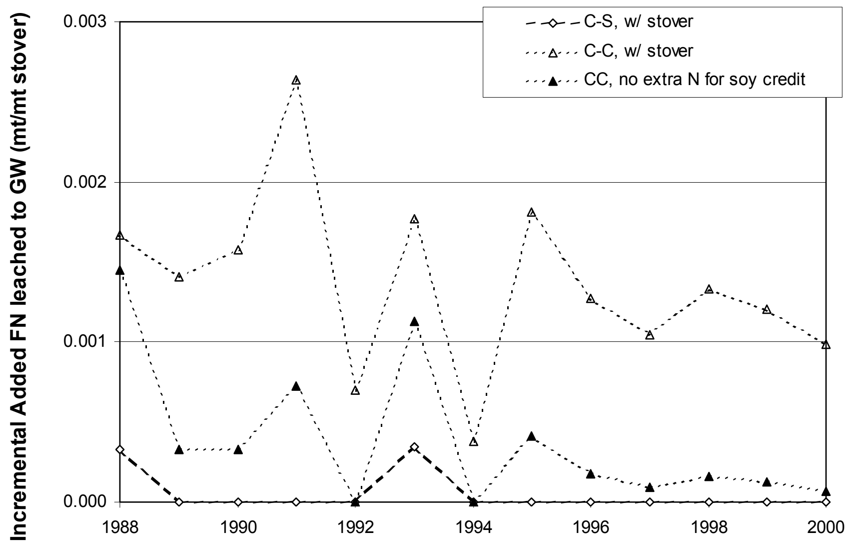


Figure 30. Incremental additional nitrogen leached to the ground water due to the collection of stover in C-C and C-S systems

The benefits in NO and N_2O emissions for the C-S with stover collection scenario are primarily the result of the switch to a no till practice. It is assumed that soil mineralization is roughly 50% more when the soil is actively broken up and exposed to oxygen during conventional till practices. With NT, the reduction in the transformation of soil organic nitrogen to ammonia and subsequently to nitrate, results in a concurrent reduction in NO and N_2O released as by products. The reduction in crop mineralization due to stover harvest also contributes some to the NO and N_2O emission benefits. These benefits of NT are also included in the C-C scenario with stover collection. In this case, however, the nitrification and denitrification associated with the much higher FN rates causes the net production of NO and N_2O to be higher than the base case.

Table 20. Incremental Percent Change in Nitrogen Emissions Due to the Harvest of Corn Stover in Comparison to the Base Case[a]

Nitrogen Flow	Scenario 2: C-S w/ stover			Scenario 3: C-C w/ stover[b]		
	average	min	max	average	min	max
(a) NH_3	1.1	0.3	2.6	28	23	36
(a) NO	-23	-41	-1.6	22	-36	90
(a) N_2O	-14	-34	8.4	60	-13	156
(w) NO_3 - SW	13	3.6	32	152	117	217
(w) NO_3-GW	46	0.0	51	92	0	676
(w) NO_3 to MSR/Gu	13	3.6	32	154	119	217

[a] only considering flows associated with soil, plants and fertilizer. Emissions associated with farm energy use not included here.
[b] assuming 45 kg/ha extra FN required for loss of soy credit

Phosphorus Flows

The phosphorus flows into and out of the two systems with stover collection are shown in Figure 31. Additional detail showing just TP leaching rates and the incremental change in these rates compared to the base case scenario are included in Figure 32. As discussed above, outflows are currently greater than inflows. The incremental amount of phosphorus that is removed with the stover harvest is very small. It does not substantially impact the need for additional FP. The most important change in these scenarios compared with the base case is related more to the switch to a no till practice from conventional till.

There is consistent and statistically significant data available that show that FP leaching rates are higher for NT practices. Based on the observations of McIsaac et al. (1995) and Gaynor and Findlay (1995), a factor of 1.6 was used here to estimate the higher leaching rates for these no till scenarios in comparison with the base case. A range of values from ~0 – 5 could be considered. This is critically important. It is the only significant cause for the observed higher TP leaching rates between the base case and the C-S scenario with stover collection. For the C-C scenario with stover collection, the greater FP usage for corn versus soy also contributes to the incremental increase in TP loads. Overall, there is less incremental increase in TP loads per harvestable stover mass in the C-C scenario versus the C-S scenario. This is the opposite of the trends observed in the TN loads between these two scenarios. The sensitivity of this result to the factor used to estimate increased TP loads in the no till scenario needs to be investigated.

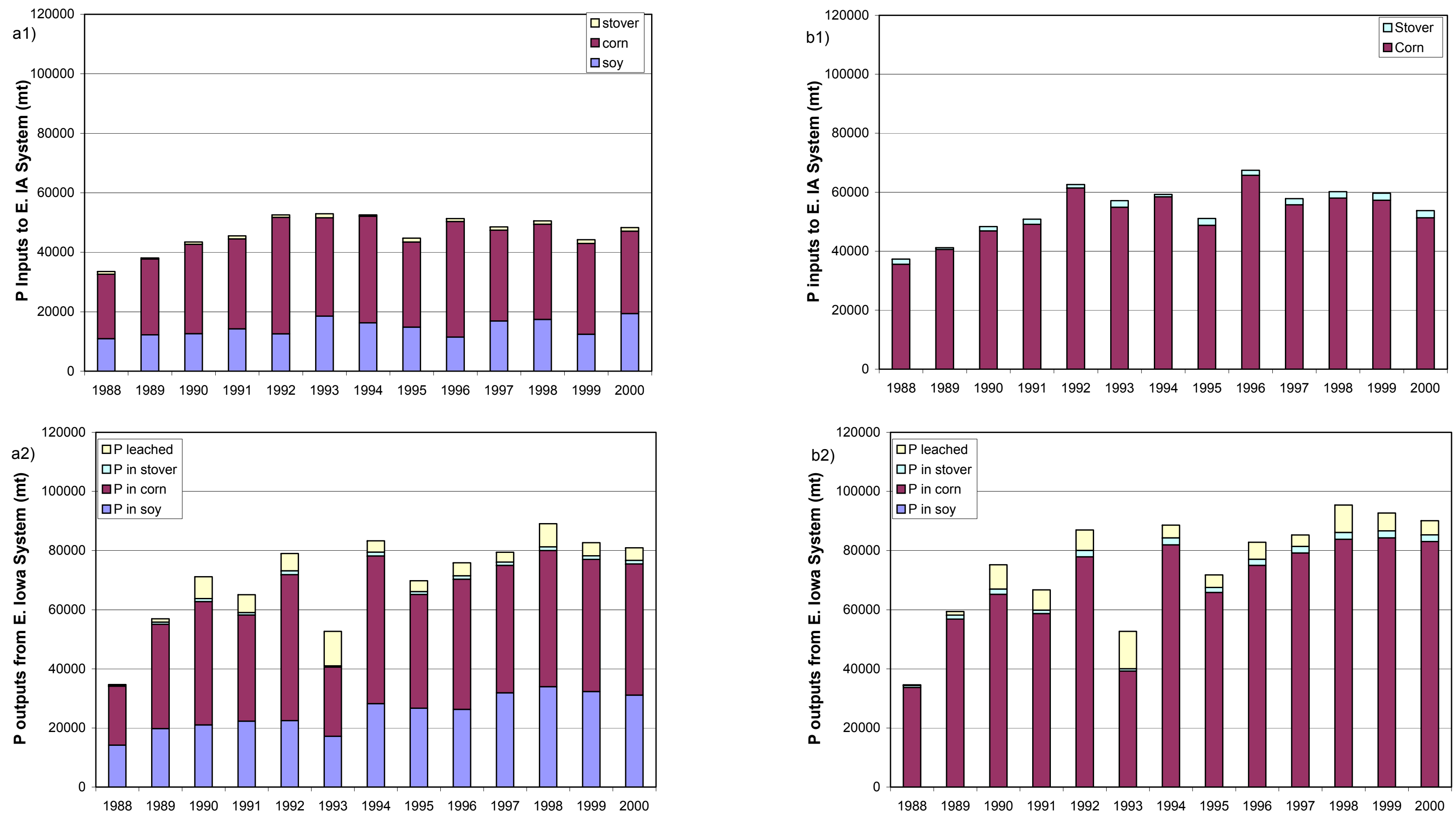


Figure 31. Phosphorus flows into (1) and out of (2) the C-S scenario with stover collection (a) and the C-C scenario with stover collection (b).

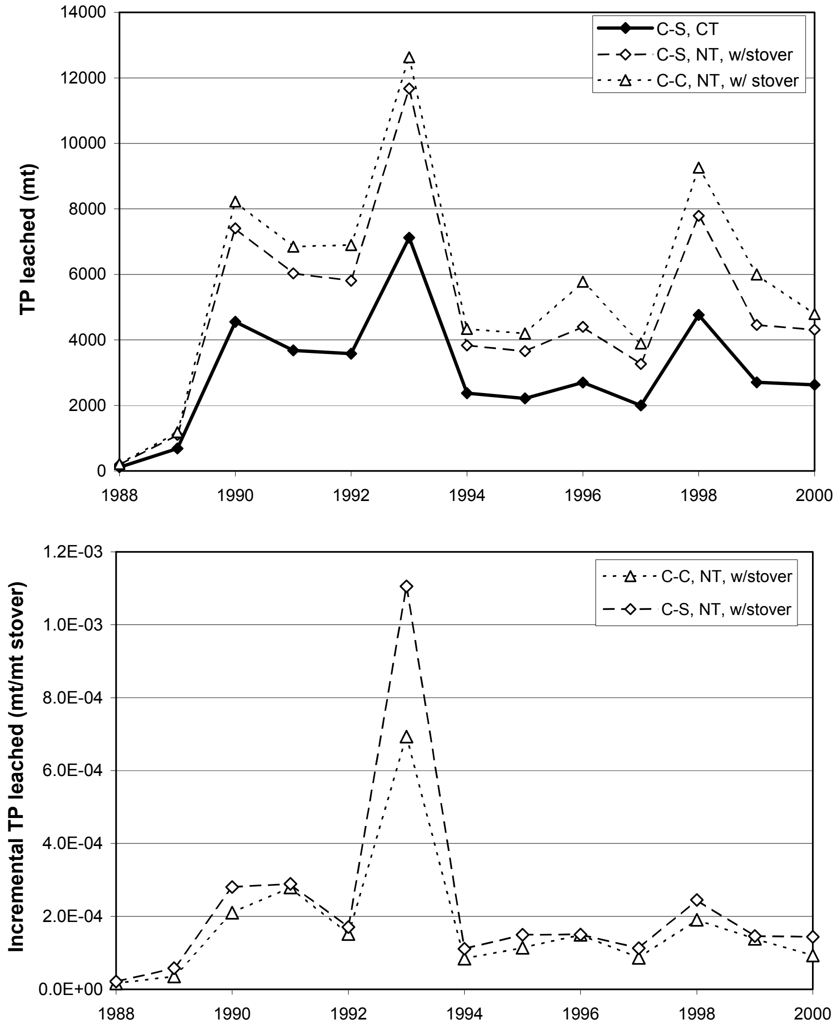


Figure 32. Phosphorus leached to the surface water for the three scenarios (top) and incremental additional TP leached due to the collection of stover in C-C and C-S systems (bottom)

Erosion

Erosion is calculated here as the total mass of sediments discharged to local surface water bodies. Unlike the nutrient models, the erosion estimates currently are based only on average rainfall data.

Figure 33 illustrates the estimated total erosion losses from C-S or C-C fields in eastern Iowa. Approximately 51% of this reaches surface water bodies and degrades water quality. Variability in estimates of erosion for the base case scenario are attributed only to variability in crop yields, not rainfall rates.

In contrast to the increase in nutrient loads associated with a switch to scenarios with stover collection, the mass of eroded soils is substantially reduced with this change. The difference is almost entirely due to the change to no till practices. The base case scenario assumes that all fields are tilled in a conventional manner. In reality, many farmers in Iowa practice some conservation tillage, so this base case is not truly representative of current conditions.

The near constant estimated erosion losses with the no till scenarios are based on the limitation of this model to include only average rainfall data. Based on an area-weighted average of the tolerable soil losses, an average of 10.8 mt soil erosion/ha is acceptable. These soil-specific tolerable soil losses were used to determine acceptable stover harvest rates each year based on an assumed average rainfall rate. The constant soil erosion shown in Figure 33 shows only the tolerable amount of erosion. It does not account for rainfall variability among years. Improvement in this model is necessary to integrate rainfall variability. The results will, however, still show substantial reductions in erosion with a switch to the stover collection scenarios with no till agricultural practices.

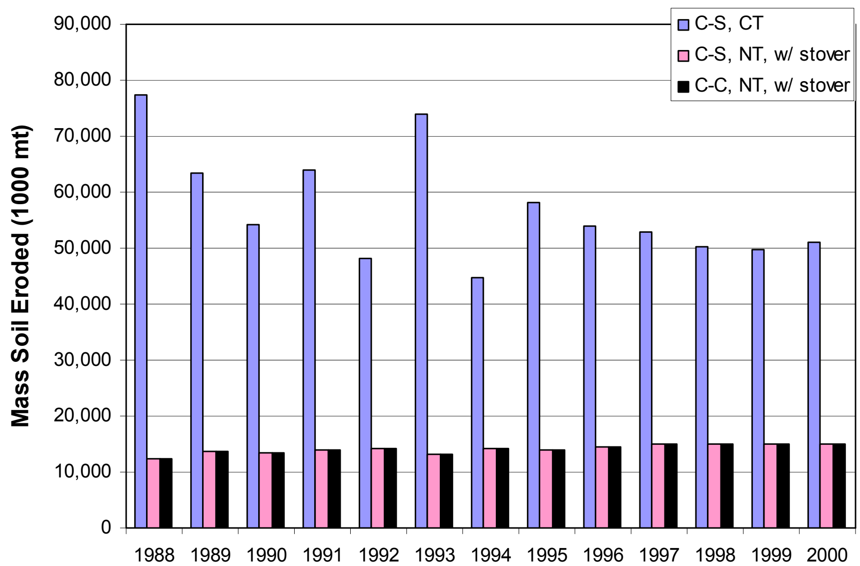


Figure 33. Soil erosion estimates for the three scenarios incorporating variability in crop yields but not rainfall.

Quantifying Flows for Life Cycle Inventory

The model generated here was used in conjunction with data from the DEAM database to quantify flows important for a lifecycle assessment of the C-S system without stover harvest and the C-C and C-S systems with stover harvest. Flows are presented as total (mt) mass flows. Normalization of these flows will depend on how they are used. Both yields (grain and stover) and land use information are included to provide data necessary for normalization at a future time.

The following tables were generated for each of the three scenarios.

- Input flows for farm processes for each of the 13 years (Tables 21-23), with averages summarized in Table 24.
- Output flows associated with nutrient use on the farm (13 years) (Tables 25-27).
- Overall LCI flows pertinent to this analysis as determined for the 13-year *average* farm inputs (Tables 28-30), with a summary and incremental changes among the scenarios in Table 31.

The average flows are aggregated by processes into five categories for each crop: "Farm," including soil and plant processes, fertilizer transformation and flows; FN production; FK+FP production; production of fossil fuels used on the farm; and, emissions from fossil fuel use on the farm.

There are a couple of key differences in the nitrogen flow numbers presented in these tables versus Figures 18, 25 and 26 due to differences in the system boundaries. The mass balance approach presented in these figures only applies to the agricultural land in the E. Iowa watershed system. Water contaminated by nitrates flows out of this system, contributing to hypoxia in the Gulf of Mexico and eventually releasing additional NO and N_2O as it eventually denitrifies. These additional emissions are included for the lifecycle inventory. Other flows (e.g., N_2 emissions), that are important for the mass balance of nitrogen flows are not important flows for the LCI and are not included in Tables 21 - 23.

Although most of the analysis of differences among the scenarios is discussed in the next section describing the impact assessment, there are a few points that should be made here to identify the various processes within the cradle-to-farm gate life cycle that contribute most to individual lifecycle flows.

Important points, Base Case Scenario:

- Ammonia releases are dominated (>99.9%) by activities associated with FN use and, more importantly, plant senescencing. The rate of senescencing used in this model is less than used by others (e.g., Goolsby et al., 1999) and there is significant uncertainty in this term.
- Carbon dioxide emissions from fossil fuels are highest for the manufacture of FN and fossil fuel consumed on the farm.
- N_2O released at the farm due to the use and transformation of FN is the only significant source of this greenhouse gas.
- Methane emissions are associated almost entirely with FN production, which uses natural gas.
- NO_x emissions are primarily the result of FN use and transformation, although fossil fuel combustion on the farm also contributes substantial amounts of NO_x.
- Fertilizer use can be considered the only source of TN and TP discharged to water bodies.
- FP production is the only significant source of acids discharged to water.
- Coal is use predominantly to generate electricity used to manufacture potassium and phosphate fertilizers.
- Oil consumption is used predominantly on the farm for planting, tilling and harvesting corn and soybeans.

Important differences among the scenarios:

- Increased fertilizer use is required for the stover collection scenarios. The incremental increase is especially high for FK because of the relatively high potassium content in stover relative to other nutrients.
- Reductions in NO and N_2O emissions in scenario 2 are related to the reduction in soil mineralization with no till practices. In scenario 3, increased use of FN and the associated nitrification is sufficient to overwhelm the benefits of reduced soil mineralization.
- Increases in the loss of total phosphorus to water bodies is the result of the assumed factor for increased TP leaching with no till practices. There is considerable uncertainty with this factor.
- Reductions in diesel fuel and its associated emissions in Scenarios 2 and 3 are the result of the no till practices.
- Substantially more fossil fuels (75%, based on total mass coal, NG and oil) are used in the overall cradle-to-farm gate generation of stover in scenario 3 – continuous corn with stover collection, versus the base case, C-S with no stover collection. The substantial increase in fertilizer use,

especially FN, contributes most to the associated increase in fossil fuel consumption. In contrast, there is only a 6.2% increase in fossil fuel consumption with scenario 2.

- Increases in TN discharges to surface water, natural gas consumption and methane generation are all most closely correlated with increased FN use.

The sensitivity of these conclusions to uncertainty in the model parameters has not yet been analyzed. Although several of the trends noted above are expected to hold, the quantitative analysis presented in the LCI tables would be better represented by ranges and distributions. Scenario 1, the corn-soybean rotation with conventional till practices is expected to be most accurate. This scenario is relatively close to actual farming practices over the period considered so that quantities of fertilizer use, plant yields and nutrient loads from the three watersheds to the Mississippi River were all historically known quantities. Mass balance verification and calibration of the nutrient leaching component of the overall model both helped to provide assurances of the adequacy of this approach and the quantitative results.

Extrapolation to the other two scenarios for which site specific historical data do not exist provides significant sources of uncertainty. Unlike Scenario 1, there are no means of verifying the results of these analyses. The quantitative analyses in these scenarios are based on a wide variety of literature sources that provide some justification for the values used. This approach inherently creates uncertainty in the model predictions due to the necessary application of parameter values measured for one system to a system with different scale, hydrology, farming practices and geology. The specific parameters for which values are most uncertain include:

- Allocation of nitrogen flows between corn and soybeans
- Ammonia released during plant senescencing
- TN and TP leaching models, especially contributions of manure to total load to the MSRB
- Fertilizer use and fate in a NT system versus convention till
- Fertilizer requirements for a C-C versus C-S system
- Actual fertilizer required to replace nutrients removed with stover
- Actual erosion rates
- Soil mineralization rates in a no till versus conventional till system.

The sensitivity of the quantitative and qualitative conclusions presented in this report to these sources of uncertainty has not been determined yet. Further research is required to identify distributions of values for these parameters that can then be used in Monte Carlo predictions of LCI results.

Table 21. Important Inputs for Farm stage of Life Cycle Inventory: C-S rotation in E. Iowa – Base case - no stover collection

Scenario 1: C-S, no stover collection

Inputs	units	Allocat	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	average
land	ha	corn	1,760,940	1,958,175	1,988,145	1,948,455	2,037,150	1,859,355	1,988,550	1,879,605	1,950,480	1,855,467	1,882,157	1,820,070	1,848,015	1,905,890
FN	mt-N	corn	271,402	277,917	271,482	256,635	258,772	231,693	265,171	241,403	280,670	249,426	257,852	253,176	257,965	259,505
FP	mt-P	corn	21,664	25,477	30,103	30,255	39,183	33,116	35,827	28,675	38,842	30,517	32,030	30,541	27,685	31,070
FK	mt-K	corn	57,540	58,966	61,998	54,459	69,735	43,162	61,138	59,211	87,644	62,572	75,938	73,019	46,591	62,459
seed	mt	corn	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568
Gasoline	mt	corn	37,659	41,877	42,518	41,669	43,566	39,764	42,527	40,197	41,712	39,681	40,251	38,924	39,521	40,759
Diesel	mt	corn	74,882	83,269	84,543	82,856	86,627	79,067	84,561	79,928	82,942	78,901	80,036	77,396	78,584	81,045
Motor oil	mt	corn	3,125	3,476	3,529	3,458	3,616	3,300	3,529	3,336	3,462	3,293	3,341	3,230	3,280	3,383
land	ha	soy	1,130,841	1,160,690	1,109,052	1,213,988	1,158,705	1,224,720	1,255,500	1,316,655	1,349,460	1,534,829	1,526,688	1,593,270	1,580,513	1,319,608
FN	mt-N	soy	2,736	1,872	4,223	5,507	3,054	1,707	2,812	4,186	3,102	4,977	3,354	3,883	13,400	4,216
FP	mt-P	soy	10,944	12,247	12,606	14,254	12,561	18,502	16,279	14,783	11,509	16,884	17,396	12,420	19,388	14,598
FK	mt-K	soy	48,983	45,925	42,927	37,591	37,709	51,781	43,300	44,958	37,722	59,802	47,263	53,557	60,494	47,078
seed	mt	soy	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526
Gasoline	mt	soy	22,330	22,919	21,900	23,972	22,880	24,184	24,791	25,999	26,647	30,307	30,146	31,461	31,209	26,057
Diesel	mt	soy	46,251	47,472	45,360	49,652	47,391	50,091	51,350	53,851	55,193	62,775	62,442	65,165	64,643	53,972
Motor oil	mt	soy	1,930	1,981	1,893	2,072	1,978	2,091	2,143	2,248	2,304	2,620	2,606	2,720	2,698	2,253
land	ha	stover														
FN	mt-N	stover														
FP	mt-P	stover														
FK	mt-K	stover														
seed	mt	stover														
Gasoline	mt	stover														
Diesel	mt	stover														
Motor oil	mt	stover														
TOTAL INPUTS																
land	ha		2,891,781	3,118,865	3,097,197	3,162,443	3,195,855	3,084,075	3,244,050	3,196,260	3,299,940	3,390,296	3,408,845	3,413,340	3,428,528	3,225,498
FN	mt-N		274,137	279,789	275,705	262,142	261,827	233,400	267,983	245,589	283,772	254,403	261,206	257,059	271,365	263,721
FP	mt-P		32,607	37,724	42,709	44,508	51,744	51,618	52,106	43,457	50,351	47,401	49,425	42,960	47,073	45,668
FK	mt-K		106,523	104,891	104,925	92,050	107,443	94,943	104,438	104,170	125,366	122,375	123,200	126,576	107,085	109,537
seed	mt		120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094
Gasoline	mt		59,989	64,796	64,417	65,641	66,446	63,947	67,318	66,196	68,359	69,987	70,397	70,384	70,730	66,816
Diesel	mt		121,133	130,741	129,904	132,508	134,018	129,158	135,911	133,779	138,135	141,676	142,478	142,561	143,228	135,018
Motor oil	mt		5,056	5,457	5,422	5,531	5,594	5,391	5,673	5,584	5,766	5,913	5,947	5,950	5,978	5,635

Scenario 1: C-S, no stover collection (cont.) method 2 - allocation of leaching based on fraction of total leaching

Inputs	units	Allocat	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	average
land	ha	corn	1,760,940	1,958,175	1,988,145	1,948,455	2,037,150	1,859,355	1,988,550	1,879,605	1,950,480	1,855,467	1,882,157	1,820,070	1,848,015	1,905,890
FN	mt-N	corn	169,794	178,525	179,751	164,226	169,544	143,162	167,053	147,030	170,734	141,999	147,058	139,880	149,231	159,076
FP	mt-P	corn	21,664	25,477	30,103	30,255	39,183	33,116	35,827	28,675	38,842	30,517	32,030	30,541	27,685	31,070
FK	mt-K	corn	57,540	58,966	61,998	54,459	69,735	43,162	61,138	59,211	87,644	62,572	75,938	73,019	46,591	62,459
seed	mt	corn	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568
Gasoline	mt	corn	37,659	41,877	42,518	41,669	43,566	39,764	42,527	40,197	41,712	39,681	40,251	38,924	39,521	40,759
Diesel	mt	corn	74,882	83,269	84,543	82,856	86,627	79,067	84,561	79,928	82,942	78,901	80,036	77,396	78,584	81,045
Motor oil	mt	corn	3,125	3,476	3,529	3,458	3,616	3,300	3,529	3,336	3,462	3,293	3,341	3,230	3,280	3,383
land	ha	soy	1,130,841	1,160,690	1,109,052	1,213,988	1,158,705	1,224,720	1,255,500	1,316,655	1,349,460	1,534,829	1,526,688	1,593,270	1,580,513	1,319,608
FN	mt-N	soy	104,344	101,263	95,954	97,916	92,283	90,238	100,930	98,560	113,038	112,404	114,149	117,178	122,135	104,645
FP	mt-P	soy	10,944	12,247	12,606	14,254	12,561	18,502	16,279	14,783	11,509	16,884	17,396	12,420	19,388	14,598
FK	mt-K	soy	48,983	45,925	42,927	37,591	37,709	51,781	43,300	44,958	37,722	59,802	47,263	53,557	60,494	47,078
seed	mt	soy	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526
Gasoline	mt	soy	22,330	22,919	21,900	23,972	22,880	24,184	24,791	25,999	26,647	30,307	30,146	31,461	31,209	26,057
Diesel	mt	soy	46,251	47,472	45,360	49,652	47,391	50,091	51,350	53,851	55,193	62,775	62,442	65,165	64,643	53,972
Motor oil	mt	soy	1,930	1,981	1,893	2,072	1,978	2,091	2,143	2,248	2,304	2,620	2,606	2,720	2,698	2,253
land	ha	stover														
FN	mt-N	stover														
FP	mt-P	stover														
FK	mt-K	stover														
seed	mt	stover														
Gasoline	mt	stover														
Diesel	mt	stover														
Motor oil	mt	stover														
TOTAL INPUTS																
land	ha		2,891,781	3,118,865	3,097,197	3,162,443	3,195,855	3,084,075	3,244,050	3,196,260	3,299,940	3,390,296	3,408,845	3,413,340	3,428,528	3,225,498
FN	mt-N		274,137	279,789	275,705	262,142	261,827	233,400	267,983	245,589	283,772	254,403	261,206	257,059	271,365	263,721
FP	mt-P		32,607	37,724	42,709	44,508	51,744	51,618	52,106	43,457	50,351	47,401	49,425	42,960	47,073	45,668
FK	mt-K		106,523	104,891	104,925	92,050	107,443	94,943	104,438	104,170	125,366	122,375	123,200	126,576	107,085	109,537
seed	mt		120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094
Gasoline	mt		59,989	64,796	64,417	65,641	66,446	63,947	67,318	66,196	68,359	69,987	70,397	70,384	70,730	66,816
Diesel	mt		121,133	130,741	129,904	132,508	134,018	129,158	135,911	133,779	138,135	141,676	142,478	142,561	143,228	135,018
Motor oil	mt		5,056	5,457	5,422	5,531	5,594	5,391	5,673	5,584	5,766	5,913	5,947	5,950	5,978	5,635

note – flows modified by the allocation method are highlighted

Table 22. Important Inputs for Farm-stage of Life Cycle Inventory: C-S rotation in E. Iowa – with stover collection

Scenario 2: C-S, NT, stover collection

Inputs	units	Allocati	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	average
land	ha	corn	1,760,940	1,958,175	1,988,145	1,948,455	2,037,150	1,859,355	1,988,550	1,879,605	1,950,480	1,855,467	1,882,157	1,820,070	1,848,015	1,905,890
FN	mt-N	corn	271,402	277,917	271,482	256,635	258,772	231,693	265,171	241,403	280,670	249,426	257,852	253,176	257,965	259,505
FP	mt-P	corn	21,664	25,477	30,103	30,255	39,183	33,116	35,827	28,675	38,842	30,517	32,030	30,541	27,685	31,070
FK	mt-K	corn	57,540	58,966	61,998	54,459	69,735	43,162	61,138	59,211	87,644	62,572	75,938	73,019	46,591	62,459
seed	mt	corn	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568	41,568
Gasoline	m^3	corn	37,659	41,877	42,518	41,669	43,566	39,764	42,527	40,197	41,712	39,681	40,251	38,924	39,521	40,759
Diesel	m^3	corn	46,380	51,575	52,364	51,319	53,655	48,972	52,375	49,505	51,372	48,869	49,572	47,937	48,673	50,197
Motor oil	m^3	corn	3,125	3,476	3,529	3,458	3,616	3,300	3,529	3,336	3,462	3,293	3,341	3,230	3,280	3,383
land	ha	soy	1,130,841	1,160,690	1,109,052	1,213,988	1,158,705	1,224,720	1,255,500	1,316,655	1,349,460	1,534,829	1,526,688	1,593,270	1,580,513	1,319,608
FN	mt-N	soy	2,736	1,872	4,223	5,507	3,054	1,707	2,812	4,186	3,102	4,977	3,354	3,883	13,400	4,216
FP	mt-P	soy	10,944	12,247	12,606	14,254	12,561	18,502	16,279	14,783	11,509	16,884	17,396	12,420	19,388	14,598
FK	mt-K	soy	48,983	45,925	42,927	37,591	37,709	51,781	43,300	44,958	37,722	59,802	47,263	53,557	60,494	47,078
seed	mt	soy	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526	78,526
Gasoline	mt	soy	22,330	22,919	21,900	23,972	22,880	24,184	24,791	25,999	26,647	30,307	30,146	31,461	31,209	26,057
Diesel	mt	soy	28,647	29,403	28,095	30,753	29,353	31,025	31,805	33,354	34,185	38,881	38,675	40,361	40,038	33,429
Motor oil	mt	soy	1,930	1,981	1,893	2,072	1,978	2,091	2,143	2,248	2,304	2,620	2,606	2,720	2,698	2,253
land	ha	stover	0	0	0	0	0	0	0	0	0	0	0	0	0	0
FN	mt-N	stover	59,332	21,870	37,356	21,948	14,292	49,000	9,146	79,237	18,957	29,127	38,513	27,609	25,952	33,257
FP	mt-P	stover	940	346	720	1,018	814	1,307	412	1,313	961	1,132	1,127	1,236	1,202	964
FK	mt-K	stover	254,394	94,649	212,345	261,590	221,884	328,114	119,122	349,309	260,814	308,519	328,860	353,833	356,663	265,392
seed	mt	stover	0	0	0	0	0	0	0	0	0	0	0	0	0	
Gasoline	mt	stover	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diesel	mt	stover	15,490	13,953	12,173	14,232	11,524	14,661	11,021	11,707	11,378	10,617	10,520	10,126	10,375	12,137
Motor oil	mt	stover	84	74	64	76	61	78	58	62	60	56	55	53	55	64
TOTAL INPUTS																
land	ha		2,891,781	3,118,865	3,097,197	3,162,443	3,195,855	3,084,075	3,244,050	3,196,260	3,299,940	3,390,296	3,408,845	3,413,340	3,428,528	3,225,498
FN	mt-N		333,470	301,659	313,061	284,090	276,119	282,399	277,129	324,826	302,730	283,530	299,720	284,667	297,318	296,978
FP	mt-P		33,547	38,070	43,429	45,527	52,558	52,925	52,518	44,770	51,312	48,533	50,553	44,197	48,275	46,632
FK	mt-K		360,917	199,540	317,270	353,640	329,328	423,057	223,560	453,478	386,180	430,893	452,060	480,410	463,748	374,929
seed	mt		120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094	120,094
Gasoline	mt		59,989	64,796	64,417	65,641	66,446	63,947	67,318	66,196	68,359	69,987	70,397	70,384	70,730	66,816
Diesel	mt		90,517	94,931	92,632	96,304	94,532	94,658	95,200	94,566	96,935	98,368	98,767	98,425	99,086	95,763
Motor oil	mt		5,140	5,531	5,486	5,606	5,654	5,469	5,731	5,646	5,825	5,969	6,002	6,004	6,033	5,700

Table 23. Important Inputs for Farm-stage of Life Cycle Inventory: C-C Rotation in E. Iowa with Stover Collection

Scenario 3: C-C, NT, stover collection

Inputs	units	Alloca	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	average
land	ha	corn	2,891,781	3,118,865	3,097,197	4,376,430	3,195,855	3,084,075	3,244,050	3,196,260	3,299,940	3,390,296	3,408,845	3,413,340	3,428,528	3,318,882
FN	mt-N	corn	575,950	583,137	562,431	773,561	549,910	523,224	578,716	554,476	623,497	608,463	620,552	628,554	633,019	601,192
FP	mt-P	corn	35,576	40,578	46,895	49,105	61,470	54,930	58,448	48,761	65,715	55,760	58,010	57,276	51,363	52,607
FK	mt-K	corn	99,418	97,163	101,258	92,806	115,613	80,054	107,620	109,947	163,186	127,059	153,795	155,201	98,793	115,532
seed	mt	corn	66,004	72,487	71,489	73,546	74,579	69,421	75,663	74,438	76,950	78,842	79,587	79,527	79,990	74,810
Gasoline	mt	corn	61,843	66,699	66,236	93,593	68,346	65,955	69,376	68,354	70,572	72,504	72,901	72,997	73,322	70,977
Diesel	mt	corn	76,164	82,145	81,574	115,267	84,173	81,229	85,442	84,183	86,914	89,294	89,782	89,901	90,301	87,413
Motor oil	mt	corn	5,133	5,536	5,497	7,768	5,672	5,474	5,758	5,673	5,857	6,017	6,050	6,058	6,085	5,891
land	ha	soy														
FN	mt-N	soy														
FP	mt-P	soy														
FK	mt-K	soy														
seed	mt	soy														
Gasoline	mt	soy														
Diesel	mt	soy														
Motor oil	mt	soy														
FN	mt-N	stover	107,057	39,831	73,462	37,588	25,968	82,011	17,624	140,568	34,300	52,924	74,905	52,684	51,338	60,789
FP	mt-P	stover	1,696	1,696	1,696	1,696	1,696	1,696	1,696	1,696	1,696	1,696	1,696	1,696	1,696	1,696
FK	mt-K	stover	254,394	254,394	254,394	254,394	254,394	254,394	254,394	254,394	254,394	254,394	254,394	254,394	254,394	254,394
seed	mt	stover	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gasoline	mt	stover	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Diesel	mt	stover	29,152	21,757	19,454	21,896	18,815	25,418	18,642	20,770	20,177	20,274	20,008	19,952	20,195	21,270
Motor oil	mt	stover	145	106	95	107	91	125	90	101	98	98	97	97	98	104
TOTAL INPUTS																
land	ha		2,891,781	3,118,865	3,097,197	4,376,430	3,195,855	3,084,075	3,244,050	3,196,260	3,299,940	3,390,296	3,408,845	3,413,340	3,428,528	3,318,882
FN	mt-N		683,006	622,968	635,893	811,149	575,878	605,235	596,340	695,044	657,798	661,388	695,456	681,238	684,358	661,981
FP	mt-P		37,272	42,274	48,591	50,801	63,166	56,626	60,144	50,457	67,411	57,456	59,706	58,972	53,059	54,303
FK	mt-K		353,812	351,558	355,653	347,200	370,008	334,449	362,015	364,342	417,581	381,454	408,190	409,596	353,187	369,926
seed	mt		66,004	72,487	71,489	73,546	74,579	69,421	75,663	74,438	76,950	78,842	79,587	79,527	79,990	74,810
Gasoline	mt		61,843	66,699	66,236	93,593	68,346	65,955	69,376	68,354	70,572	72,504	72,901	72,997	73,322	70,977
Diesel	mt		105,316	103,902	101,029	137,163	102,988	106,646	104,085	104,953	107,091	109,568	109,791	109,853	110,496	108,683
Motor oil	mt		5,278	5,642	5,592	7,875	5,764	5,599	5,848	5,774	5,955	6,116	6,147	6,155	6,183	5,994

Table 24. Summary of Average[a] Farm Inputs and Incremental Percentage Differences among Scenarios

Scenario 1: Base Case, C-S, no stover (allocation method 1)

	(avg. mt)			
	Corn	Soy	Stover	TOTAL
FN	259,505	4,216		263,721
FP	31,070	14,598		45,668
FK	62,459	47,078		109,537
Gasoline	40,759	26,057		66,816
Diesel	81,045	53,972		135,018
Motor oil	3,383	2,253		5,635

Scenario 2: C-S, w/ stover (allocation method 1)

	(avg. mt)				Incremental % Change compared with Base Case			
	Corn	Soy	Stover	TOTAL	Corn	Soy	Stover	TOTAL
FN	259,505	4,216	33,257	296,978	0	0	100	12.6
FP	31,070	14,598	964	46,632	0	0	100	2.1
FK	62,459	47,078	265,392	374,929	0	0	100	242.3
Gasoline	40,759	26,057	0	66,816	0	0	100	0.0
Diesel	50,197	33,429	12,137	95,763	-38.06	-38.06	100	-29.1
Motor oil	3,383	2,253	64	5,700	0	0	100	1.1

Scenario 3: C-C, w/stover

	(avg. mt)				Incremental % Change compared with Base Case			
	Corn	Soy	Stover	TOTAL	Corn	Soy	Stover	TOTAL
FN	601,192		60,789	661,981	131.7	-100	100	151.0
FP	52,607		1,696	54,303	69.3	-100	100	18.9
FK	115,532		254,394	369,926	85.0	-100	100	237.7
Gasoline	70,977		0	70,977	74.1	-100	100	6.2
Diesel	87,413		21,270	108,683	7.9	-100	100	-19.5
Motor oil	5,891		104	5,994	74.1	-100	100	6.4

[a] Averaged over 13 year period (from Tables 21-23)

Table 25. Important Nutrient Outflows for Farm stage of Life Cycle Inventory: C-S rotation in E. Iowa – Scenario 1: Base case - no stover collection

Scenario 1: C-S, no stover collection Method 1: Allocation of N leaching to crop FN applied

Outputs	units	Allocati	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	average
grain	mt	corn	7,462,069	13,212,182	15,602,329	13,434,921	18,448,394	8,763,835	18,677,006	14,388,681	16,469,321	16,129,678	17,223,270	16,704,426	16,583,082	14,853,784
erosion	1000 mt-	corn	47,100	39,848	34,879	39,374	30,625	44,518	27,354	34,247	31,818	28,913	27,767	26,466	27,586	33,884
(a) NH3	mt-N	corn	58,352	65,688	66,173	65,041	67,906	59,329	66,559	64,665	65,843	62,079	63,279	61,072	62,066	63,696
(a) NO	mt-N	corn	1,974	2,932	5,291	4,311	4,426	5,936	4,166	3,345	3,843	3,349	4,833	3,755	3,925	4,006
(a) N2O	mt-N	corn	1,153	1,890	4,326	3,421	3,310	5,388	2,898	2,484	2,904	2,396	3,875	2,868	2,877	3,061
(w) TN - SW	mt-N	corn	16,594	45,101	182,646	137,595	119,271	196,841	86,694	86,304	104,548	76,743	158,488	107,679	99,429	109,072
(w) NO3-GW	mt-N	corn	2,457	0	0	0	0	1,924	0	0	0	0	0	0	0	337
(w) TN to MSR/Gu	mt-N	corn	9,956	27,060	146,117	110,076	95,417	157,473	52,017	51,782	62,729	46,046	126,790	86,143	79,543	80,858
(w) TP-SW	mt-P	corn	66	435	3,045	2,269	2,477	4,110	1,521	1,312	1,695	1,148	2,737	1,576	1,547	1,841
grain	mt	soy	2,175,206	3,025,679	3,218,404	3,415,251	3,450,599	2,630,838	4,323,526	4,085,288	4,027,441	4,879,986	5,203,268	4,949,030	4,769,339	3,857,989
erosion	1000 mt-	soy	30,247	23,620	19,456	24,532	17,419	29,323	17,271	23,990	22,014	23,917	22,523	23,168	23,593	23,159
(a) NH3	mt-N	soy	22,530	23,238	22,010	24,249	23,184	23,949	25,121	27,039	27,007	30,551	30,502	31,844	31,956	26,398
(a) NO	mt-N	soy	815	939	977	1,059	1,008	1,105	1,152	1,163	1,166	1,362	1,411	1,405	1,442	1,154
(a) N2O	mt-N	soy	409	472	514	558	517	742	585	596	594	695	725	718	768	607
(w) TN - SW	mt-N	soy	172	299	2,820	3,081	1,388	1,456	925	1,518	1,151	1,527	2,052	1,645	5,127	1,782
(w) NO3-GW	mt-N	soy	25	0	0	0	0	14	0	0	0	0	0	0	0	3
(w) NO3 to MSR/Gu	mt-N	soy	103	180	2,256	2,465	1,111	1,165	555	911	691	916	1,642	1,316	4,102	1,339
(w) TP-SW	mt-P	soy	45	250	1,511	1,412	1,104	3,012	855	906	1,005	850	2,027	1,128	1,083	1,168
grain	mt	stover														
erosion	1000 mt-	stover														
(a) NH3	mt-N	stover														
(a) NO	mt-N	stover														
(a) N2O	mt-N	stover														
(w) TN - SW	mt-N	stover														
(w) NO3-GW	mt-N	stover														
(w) TN to MSR/Gu	mt-N	stover														
(w) TP-SW	mt-P	stover														
TOTAL OUTFLOWS																
grain	mt	corn	7,462,069	13,212,182	15,602,329	13,434,921	18,448,394	8,763,835	18,677,006	14,388,681	16,469,321	16,129,678	17,223,270	16,704,426	16,583,082	14,853,784
grain	mt	soy	2,175,206	3,025,679	3,218,404	3,415,251	3,450,599	2,630,838	4,323,526	4,085,288	4,027,441	4,879,986	5,203,268	4,949,030	4,769,339	3,857,989
grain	mt	stover	0	0	0	0	0	0	0	0	0	0	0	0	0	0
erosion	1000 mt-soil		77,347	63,468	54,335	63,906	48,043	73,842	44,625	58,237	53,832	52,830	50,290	49,635	51,179	57,044
(a) NH3	mt-N		80,882	88,925	88,183	89,290	91,090	83,279	91,680	91,705	92,850	92,630	93,781	92,916	94,022	90,095
(a) NO	mt-N		2,789	3,871	6,268	5,370	5,434	7,041	5,318	4,509	5,009	4,711	6,245	5,160	5,366	5,161
(a) N2O	mt-N		1,562	2,362	4,840	3,979	3,827	6,130	3,483	3,080	3,498	3,091	4,599	3,586	3,645	3,668
(w) TN - SW	mt-N		16,766	45,400	185,466	140,675	120,659	198,297	87,620	87,822	105,699	78,270	160,540	109,324	104,556	110,853
(w) NO3-GW	mt-N		2,482	0	0	0	0	1,938	0	0	0	0	0	0	0	340
(w) TN to MSR/Gu	mt-N		10,060	27,240	148,372	112,540	96,527	158,638	52,572	52,693	63,420	46,962	128,432	87,459	83,645	82,197
(w) TP-SW	mt-P		112	685	4,556	3,681	3,581	7,122	2,376	2,218	2,700	1,998	4,764	2,704	2,630	3,010

Scenario 1: C-S,no stover collection (cont.) method 2 - allocation of leaching based on fraction of total leaching

Outputs	units	Allocati	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
grain	mt	corn	7,462,069	13,212,182	15,602,329	13,434,921	18,448,394	8,763,835	18,677,006	14,388,681	16,469,321	16,129,678	17,223,270	16,704,426	16,583,082
erosion	1000 mt-s	corn	47,100	39,848	34,879	39,374	30,625	44,518	27,354	34,247	31,818	28,913	27,767	26,466	27,586
(a) NH3	mt-N	corn	58,352	65,688	66,173	65,041	67,906	59,329	66,559	64,665	65,843	62,079	63,279	61,072	62,066
(a) NO	mt-N	corn	1,856	2,627	4,136	3,386	3,657	4,512	3,560	2,708	3,069	2,724	3,560	2,854	3,140
(a) N2O	mt-N	corn	1,036	1,586	3,181	2,504	2,547	3,967	2,293	1,848	2,132	1,773	2,611	1,974	2,099
(w) TN - SW	mt-N	corn	10,385	28,968	120,918	88,130	78,132	121,631	54,620	52,577	63,595	43,688	90,383	59,489	57,498
(w) NO3-GW	mt-N	corn	2,457	0	0	0	0	1,924	0	0	0	0	0	0	0
(w) TN to MSR/Gu	mt-N	corn	6,231	17,381	96,734	70,504	62,506	97,305	32,772	31,546	38,157	26,213	72,306	47,591	45,999
(w) TP-SW	mt-P	corn	66	435	3,045	2,269	2,477	4,110	1,521	1,312	1,695	1,148	2,737	1,576	1,547
grain	mt	soy	2,175,206	3,025,679	3,218,404	3,415,251	3,450,599	2,630,838	4,323,526	4,085,288	4,027,441	4,879,986	5,203,268	4,949,030	4,769,339
erosion	1000 mt-s	soy	30,247	23,620	19,456	24,532	17,419	29,323	17,271	23,990	22,014	23,917	22,523	23,168	23,593
(a) NH3	mt-N	soy	22,530	23,238	22,010	24,249	23,184	23,949	25,121	27,039	27,007	30,551	30,502	31,844	31,956
(a) NO	mt-N	soy	933	1,244	2,131	1,984	1,777	2,529	1,759	1,801	1,940	1,987	2,685	2,306	2,226
(a) N2O	mt-N	soy	526	776	1,659	1,475	1,280	2,163	1,190	1,232	1,366	1,319	1,988	1,611	1,546
(w) TN - SW	mt-N	soy	6,382	16,431	64,548	52,545	42,527	76,667	33,000	35,245	42,105	34,582	70,157	49,834	47,058
(w) NO3-GW	mt-N	soy	25	0	0	0	0	14	0	0	0	0	0	0	0
(w) TN to MSR/Gu	mt-N	soy	3,829	9,859	51,638	42,036	34,022	61,333	19,800	21,147	25,263	20,749	56,125	39,867	37,647
(w) TP-SW	mt-P	soy	45	250	1,511	1,412	1,104	3,012	855	906	1,005	850	2,027	1,128	1,083
grain	mt	stover													
erosion	1000 mt-s	stover													
(a) NH3	mt-N	stover													
(a) NO	mt-N	stover													
(a) N2O	mt-N	stover													
(w) TN - SW	mt-N	stover													
(w) NO3-GW	mt-N	stover													
(w) TN to MSR/Gu	mt-N	stover													
(w) TP-SW	mt-P	stover													
TOTAL OUTFLOWS															
grain	mt	corn	7,462,069	13,212,182	15,602,329	13,434,921	18,448,394	8,763,835	18,677,006	14,388,681	16,469,321	16,129,678	17,223,270	16,704,426	16,583,082
grain	mt	soy	2,175,206	3,025,679	3,218,404	3,415,251	3,450,599	2,630,838	4,323,526	4,085,288	4,027,441	4,879,986	5,203,268	4,949,030	4,769,339
grain	mt	stover	0	0	0	0	0	0	0	0	0	0	0	0	0
erosion	1000 mt-soil		77,347	63,468	54,335	63,906	48,043	73,842	44,625	58,237	53,832	52,830	50,290	49,635	51,179
(a) NH3	mt-N		80,882	88,925	88,183	89,290	91,090	83,279	91,680	91,705	92,850	92,630	93,781	92,916	94,022
(a) NO	mt-N		2,789	3,871	6,268	5,370	5,434	7,041	5,318	4,509	5,009	4,711	6,245	5,160	5,366
(a) N2O	mt-N		1,562	2,362	4,840	3,979	3,827	6,130	3,483	3,080	3,498	3,091	4,599	3,586	3,645
(w) TN - SW	mt-N		16,766	45,400	185,466	140,675	120,659	198,297	87,620	87,822	105,699	78,270	160,540	109,324	104,556
(w) NO3-GW	mt-N		2,482	0	0	0	0	1,938	0	0	0	0	0	0	0
(w) TN to MSR/Gu	mt-N		10,060	27,240	148,372	112,540	96,527	158,638	52,572	52,693	63,420	46,962	128,432	87,459	83,645
(w) TP-SW	mt-P		112	685	4,556	3,681	3,581	7,122	2,376	2,218	2,700	1,998	4,764	2,704	2,630

note – flows modified by the allocation method are highlighted

Table 26. Important Nutrient Outflows for Farm Stage of Life Cycle Inventory: Scenario 2: C-S rotation in E. Iowa with Stover Collection

Scenario 2: C-S, NT, stover collection (C-S allocation by method 1)

Outputs	units	Allocat	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	average
grain	mt	corn	7,462,069	13,212,182	15,602,329	13,434,921	18,448,394	8,763,835	18,677,006	14,388,681	16,469,321	16,129,678	17,223,270	16,704,426	16,583,082	14,853,784
erosion	1000 mt-s	corn	12,182	12,499	11,944	13,077	12,482	13,192	13,522	14,170	14,531	16,533	16,444	17,152	17,017	14,211
(a) NH3	mt-N	corn	58,352	65,688	66,173	65,041	67,906	59,329	66,559	64,665	65,843	62,079	63,279	61,072	62,066	63,696
(a) NO	mt-N	corn	1,974	2,932	5,291	4,311	4,426	5,936	4,166	3,345	3,843	3,349	4,833	3,755	3,925	4,006
(a) N2O	mt-N	corn	1,153	1,890	4,326	3,421	3,310	5,388	2,898	2,484	2,904	2,396	3,875	2,868	2,877	3,061
(w) TN - SW	mt-N	corn	16,594	45,101	182,646	137,595	119,271	196,841	86,694	86,304	104,548	76,743	158,488	107,679	99,429	109,072
(w) NO3-GW	mt-N	corn	2,457	0	0	0	0	1,924	0	0	0	0	0	0	0	337
(w) TN to MSR/Gu	mt-N	corn	9,956	27,060	146,117	110,076	95,417	157,473	52,017	51,782	62,729	46,046	126,790	86,143	79,543	80,858
(w) TP-SW	mt-P	corn	66	435	3,045	2,269	2,477	4,110	1,521	1,312	1,695	1,148	2,737	1,576	1,547	1,841
grain	mt	soy	2,175,206	3,025,679	3,218,404	3,415,251	3,450,599	2,630,838	4,323,526	4,085,288	4,027,441	4,879,986	5,203,268	4,949,030	4,769,339	3,857,989
erosion	1000 mt-s	soy	18,951	21,077	21,400	20,967	21,925	20,014	21,404	20,252	20,994	19,973	20,258	19,591	19,889	20,515
(a) NH3	mt-N	soy	22,530	23,238	22,010	24,249	23,184	23,949	25,121	27,039	27,007	30,551	30,502	31,844	31,956	26,398
(a) NO	mt-N	soy	815	939	977	1,059	1,008	1,105	1,152	1,163	1,166	1,362	1,411	1,405	1,442	1,154
(a) N2O	mt-N	soy	409	472	514	558	517	742	585	596	594	695	725	718	768	607
(w) TN - SW	mt-N	soy	172	299	2,820	3,081	1,388	1,456	925	1,518	1,151	1,527	2,052	1,645	5,127	1,782
(w) NO3-GW	mt-N	soy	25	0	0	0	0	14	0	0	0	0	0	0	0	3
(w) TN to MSR/Gu	mt-N	soy	103	180	2,256	2,465	1,111	1,165	555	911	691	916	1,642	1,316	4,102	1,339
(w) TP-SW	mt-P	soy	45	250	1,511	1,412	1,104	3,012	855	906	1,005	850	2,027	1,128	1,083	1,168
grain	mt	stover	3,464,615	7,198,571	10,183,065	8,141,262	13,069,415	4,121,022	13,126,850	9,609,870	11,319,570	11,272,459	12,361,503	12,019,927	11,743,721	9,817,835
erosion	1000 mt-s	stover	0	0	0	0	0	0	0	0	0	0	0	0	0	0
(a) NH3	mt-N	stover	1,780	656	1,121	658	429	1,470	274	2,377	569	874	1,155	828	779	998
(a) NO	mt-N	stover	-1,066	-1,343	-676	-945	-1,403	-111	-1,741	-712	-1,221	-1,291	-934	-1,106	-1,390	-1,072
(a) N2O	mt-N	stover	-490	-654	-17	-358	-645	521	-841	-94	-546	-565	-255	-439	-603	-384
(w) TN - SW	mt-N	stover	4,057	1,894	24,861	12,440	6,099	40,733	3,179	27,874	6,885	8,527	23,060	12,427	9,975	14,001
(w) NO3-GW	mt-N	stover	1,144	0	0	0	0	1,204	0	0	0	0	0	0	0	181
(w) TN to MSR/Gu	mt-N	stover	2,434	1,136	19,889	9,952	4,879	32,586	1,907	16,725	4,131	5,116	18,448	9,942	7,980	10,394
(w) TP-SW	mt-P	stover	73	415	2,855	2,352	2,231	4,555	1,458	1,436	1,700	1,271	3,027	1,756	1,685	1,909
TOTAL OUTFLOWS																
grain	mt	corn	7,462,069	13,212,182	15,602,329	13,434,921	18,448,394	8,763,835	18,677,006	14,388,681	16,469,321	16,129,678	17,223,270	16,704,426	16,583,082	14,853,784
grain	mt	soy	2,175,206	3,025,679	3,218,404	3,415,251	3,450,599	2,630,838	4,323,526	4,085,288	4,027,441	4,879,986	5,203,268	4,949,030	4,769,339	3,857,989
grain	mt	stover	3,464,615	7,198,571	10,183,065	8,141,262	13,069,415	4,121,022	13,126,850	9,609,870	11,319,570	11,272,459	12,361,503	12,019,927	11,743,721	9,817,835
erosion	1000 mt-soil		31,133	33,576	33,344	34,044	34,407	33,206	34,927	34,422	35,525	36,506	36,703	36,744	36,906	34,726
(a) NH3	mt-N		82,662	89,582	89,304	89,948	91,518	84,749	91,954	94,082	93,419	93,504	94,936	93,745	94,801	91,093
(a) NO	mt-N		1,723	2,528	5,592	4,425	4,031	6,930	3,577	3,796	3,788	3,420	5,311	4,054	3,976	4,089
(a) N2O	mt-N		1,072	1,708	4,823	3,621	3,182	6,650	2,642	2,986	2,952	2,526	4,344	3,147	3,042	3,284
(w) TN - SW	mt-N		20,823	47,294	210,327	153,115	126,758	239,030	90,799	115,696	112,585	86,797	183,599	121,751	114,531	124,854
(w) NO3-GW	mt-N		3,626	0	0	0	0	3,142	0	0	0	0	0	0	0	521
(w) TN to MSR/Gu	mt-N		12,494	28,376	168,261	122,492	101,406	191,224	54,479	69,418	67,551	52,078	146,879	97,401	91,625	92,591
(w) TP-SW	mt-P		185	1,100	7,410	6,033	5,811	11,677	3,835	3,654	4,400	3,269	7,791	4,460	4,315	4,918

Table 27. Important Nutrient Outflows for Farm Stage of Life Cycle Inventory: Scenario 3: C-C rotation in E. Iowa with Stover Collection

Scenario 3: C-C, NT, stover collection

Outputs	units	Allocati	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	average
grain	mt	corn	12,619,734	21,227,138	24,395,565	21,955,518	29,128,265	14,681,650	30,635,079	24,605,727	28,036,781	29,602,371	31,326,206	31,516,110	31,034,408	25,443,427
erosion	1000 mt-s	corn	31,133	33,576	33,344	34,044	34,407	33,206	34,927	34,422	35,525	36,506	36,703	36,744	36,906	34,726
(a) NH3	mt-N	corn	88,013	103,727	97,231	112,343	107,004	91,968	110,678	94,650	111,400	110,435	108,868	112,109	113,022	104,727
(a) NO	mt-N	corn	1,508	2,753	8,532	9,715	5,707	10,958	4,719	4,504	5,304	4,467	8,233	5,907	5,578	5,991
(a) N2O	mt-N	corn	1,143	2,277	8,391	9,732	5,187	11,398	4,152	4,162	4,862	4,013	7,831	5,441	5,049	5,664
(w) TN - SW	mt-N	corn	35,770	92,292	377,394	424,840	251,782	445,031	189,681	199,361	231,801	186,657	380,309	267,330	243,148	255,800
(w) NO3-GW	mt-N	corn	13,079	8,287	8,798	15,851	4,198	13,008	2,337	9,136	7,620	6,085	7,509	6,995	5,759	8,359
(w) TN to MSR/Gu	mt-N	corn	21,462	55,375	301,915	339,872	201,426	356,025	113,809	119,617	139,080	111,994	304,247	213,864	194,519	190,247
(w) TP-SW	mt-P	corn	203	1,173	7,981	6,610	6,772	12,146	4,277	4,007	5,629	3,758	8,929	5,757	4,577	5,525
grain	mt	soy														
erosion	1000 mt-s	soy														
(a) NH3	mt-N	soy														
(a) NO	mt-N	soy														
(a) N2O	mt-N	soy														
(w) TN - SW	mt-N	soy														
(w) NO3-GW	mt-N	soy														
(w) TN to MSR/Gu	mt-N	soy														
(w) TP-SW	mt-P	soy														
grain	mt	stover	6,309,940	14,156,331	17,439,362	14,792,296	21,874,281	7,941,495	23,287,245	17,387,602	20,567,908	21,924,007	23,588,884	23,777,544	23,246,085	18,176,383
erosion	1000 mt-s	stover	0	0	0	0	0	0	0	0	0	0	0	0	0	0
(a) NH3	mt-N	stover	16,360	7,085	12,700	5,459	5,053	14,415	3,371	23,995	6,128	9,606	13,141	9,397	9,166	10,452
(a) NO	mt-N	stover	280	188	1,114	472	270	1,718	144	1,142	292	389	994	495	452	611
(a) N2O	mt-N	stover	212	156	1,096	473	245	1,786	126	1,055	267	349	945	456	409	583
(w) TN - SW	mt-N	stover	6,649	6,304	49,293	20,644	11,890	69,755	5,777	50,541	12,752	16,236	45,906	22,407	19,720	25,990
(w) NO3-GW	mt-N	stover	2,431	566	1,149	770	198	2,039	71	2,316	419	529	906	586	467	958
(w) TN to MSR/Gu	mt-N	stover	3,989	3,782	39,434	16,515	9,512	55,804	3,466	30,325	7,651	9,741	36,725	17,926	15,776	19,280
(w) TP-SW	mt-P	stover	10	11	241	238	125	481	58	191	147	135	334	244	212	187
TOTAL OUTFLOWS																
grain	mt	corn	12,619,734	21,227,138	24,395,565	21,955,518	29,128,265	14,681,650	30,635,079	24,605,727	28,036,781	29,602,371	31,326,206	31,516,110	31,034,408	25,443,427
grain	mt	soy	0	0	0	0	0	0	0	0	0	0	0	0	0	0
grain	mt	stover	6,309,940	14,156,331	17,439,362	14,792,296	21,874,281	7,941,495	23,287,245	17,387,602	20,567,908	21,924,007	23,588,884	23,777,544	23,246,085	18,176,383
erosion	1000 mt-soil		31,133	33,576	33,344	34,044	34,407	33,206	34,927	34,422	35,525	36,506	36,703	36,744	36,906	34,726
(a) NH3	mt-N		104,373	110,812	109,931	117,802	112,057	106,383	114,049	118,645	117,528	120,040	122,009	121,506	122,189	115,179
(a) NO	mt-N		1,789	2,941	9,647	10,187	5,977	12,675	4,862	5,645	5,596	4,856	9,227	6,402	6,030	6,603
(a) N2O	mt-N		1,355	2,433	9,487	10,205	5,432	13,184	4,278	5,218	5,129	4,362	8,776	5,897	5,458	6,247
(w) TN - SW	mt-N		42,419	98,596	426,687	445,483	263,672	514,786	195,458	249,902	244,553	202,893	426,215	289,738	262,868	281,790
(w) NO3-GW	mt-N		15,510	8,853	9,947	16,621	4,396	15,047	2,408	11,452	8,039	6,614	8,415	7,582	6,227	9,316
(w) TN to MSR/Gu	mt-N		25,452	59,158	341,350	356,387	210,938	411,829	117,275	149,941	146,732	121,736	340,972	231,790	210,294	209,527
(w) TP-SW	mt-P		214	1,184	8,222	6,848	6,896	12,628	4,335	4,198	5,777	3,892	9,263	6,001	4,789	5,711

Table 28. Overall Annual Cradle-to-Farm Gate LCI Flows and Distributions among Processes, Scenario 1: Base Case, C-S, no Stover Collection[a]

(Scenario 1: Base case, C-S allocation method 1)

Cradle to Farmgate Lifecycle Flows (in mt unless noted)	Corn Farm	Corn FN prod	Corn FP+FK prod	Corn energy prod	Corn energy farm	Soy Farm	Soy FN prod	Soy FP+FK prod	Soy energy prod	Soy energy farm	TOTAL
(a) Ammonia (NH3-N)	6.37E+04	2.60E-01	1.33E-02	2.79E-01	0.00E+00	2.64E+04	4.22E-03	7.46E-03	1.86E-01	0.00E+00	9.01E+04
(a) Carbon Dioxide (CO2, fossil)		6.62E+05	1.01E+05	5.00E+04	3.89E+05		1.08E+04	5.54E+04	3.31E+04	2.56E+05	1.56E+06
(a) Carbon Monoxide (CO)		1.58E+01	4.79E+01	3.63E+01	1.47E+03		2.56E-01	2.75E+01	2.41E+01	9.71E+02	2.59E+03
(a) Methane (CH4)		5.36E+03	1.95E+02	1.08E+02	0.00E+00		8.70E+01	1.13E+02	7.18E+01	0.00E+00	5.93E+03
(a) Nitrogen Oxides (NOx as NO2)	4.01E+03	8.82E+02	8.98E+00	1.40E+02	3.15E+03	1.15E+03	1.43E+01	2.13E+02	9.28E+01	2.09E+03	1.18E+04
(a) Nitrous Oxide (N2O)	3.06E+03	0.00E+00	8.98E+00	4.57E+00	6.70E+00	6.07E+02	0.00E+00	5.09E+00	3.01E+00	4.46E+00	3.70E+03
(a) Sulfur Oxides (SOx as SO2)		3.15E+02	1.90E+03	4.16E+02	4.64E+02		5.12E+00	9.20E+02	2.76E+02	2.61E+01	4.32E+03
(w) Acids (H+)		1.58E-01	4.73E+03	1.04E-01	0.00E+00		2.56E-03	2.22E+03	6.90E-02	0.00E+00	6.95E+03
(w) TN	1.09E+05	3.15E-01	2.73E-01	1.60E+01	0.00E+00	1.78E+03	5.12E-03	1.68E-01	1.05E+01	0.00E+00	1.11E+05
(w) TP	1.84E+03	0.00E+00	5.65E-01	1.94E-02	0.00E+00	1.17E+03	0.00E+00	2.66E-01	1.24E-02	0.00E+00	3.01E+03
(r) Land (ha)	1.91E+06					1.32E+06					3.23E+06
(r) Coal (in ground)		7.54E+03	2.53E+04	4.23E+03			1.23E+02	1.42E+04	2.78E+03		5.42E+04
(r) Natural Gas (in ground)		2.39E+05	4.32E+04	1.16E+04			3.89E+03	2.09E+04	7.61E+03		3.26E+05
(r) Oil (in ground)		9.80E+02	1.11E+04	9.29E+04			1.59E+01	5.56E+03	6.19E+04		1.72E+05
(r) Potassium Chloride (KCl, as K2O, in ground)			6.37E+04					4.80E+04			1.12E+05
(r) Phosphate Rock (in ground)			2.31E+05					1.09E+05			3.40E+05
Water Used (total) (m^3)		3.47E+06	2.67E+05	4.49E+05			5.63E+04	1.28E+05	2.93E+05		4.66E+06

% of total	Corn Farm	Corn FN prod	Corn FP+FK prod	Corn energy prod	Corn energy farm	Soy Farm	Soy FN prod	Soy FP+FK prod	Soy energy prod	Soy energy farm
(a) Ammonia (NH3-N)	70.7	0.0	0.0	0.0	0.0	29.3	0.0	0.0	0.0	0.0
(a) Carbon Dioxide (CO2, fossil)		42.5	6.5	3.2	25.0		0.7	3.6	2.1	16.4
(a) Carbon Monoxide (CO)		0.6	1.8	1.4	56.7		0.0	1.1	0.9	37.4
(a) Methane (CH4)		90.3	3.3	1.8	0.0		1.5	1.9	1.2	0.0
(a) Nitrogen Oxides (NOx as NO2)	34.1	7.5	0.1	1.2	26.8	9.8	0.1	1.8	0.8	17.8
(a) Nitrous Oxide (N2O)	82.7	0.0	0.2	0.1	0.2	16.4	0.0	0.1	0.1	0.1
(a) Sulfur Oxides (SOx as SO2)		7.3	44.0	9.6	10.7		0.1	21.3	6.4	0.6
(w) Acids (H+)		0.0	68.0	0.0	0.0		0.0	32.0	0.0	0.0
(w) TN	98.4	0.0	0.0	0.0	0.0	1.6	0.0	0.0	0.0	0.0
(w) TP	61.2	0.0	0.0	0.0	0.0	38.8	0.0	0.0	0.0	0.0
(r) Land (ha)	59.1	0.0	0.0	0.0		40.9	0.0	0.0	0.0	
(r) Coal (in ground)		13.9	46.8	7.8			0.2	26.2	5.1	
(r) Natural Gas (in ground)		73.3	13.2	3.5			1.2	6.4	2.3	
(r) Oil (in ground)		0.6	6.5	53.9			0.0	3.2	35.9	
(r) Potassium Chloride (KCl, as K2O, in ground)			57.0					43.0		
(r) Phosphate Rock (in ground)			68.0					32.0		
Water Used (total) (m^3)		74.4	5.7	9.6			1.2	2.7	6.3	

[a] using average of 13 year inflow data (Table 24)

Table 28 (cont.). Overall Annual Cradle-to-Farm Gate LCI Flows and Distributions among Processes, Scenario 1: Base Case, C-S, no Stover Collection

(Scenario 1: Base case, C-S allocation method 2)

Cradle to Farmgate Lifecycle Flows (in mt unless noted)	Corn Farm	Corn FN prod	Corn FP+FK prod	Corn energy prod	Corn energy farm	Soy Farm	Soy FN prod	Soy FP+FK prod	Soy energy prod	Soy energy farm	TOTAL
(a) Ammonia (NH3-N)	6.37E+04	1.59E-01	1.33E-02	2.79E-01	0.00E+00	2.64E+04	1.05E-01	7.46E-03	1.86E-01	0.00E+00	9.01E+04
(a) Carbon Dioxide (CO2, fossil)		4.06E+05	1.01E+05	5.00E+04	3.89E+05		2.67E+05	5.54E+04	3.31E+04	2.56E+05	1.56E+06
(a) Carbon Monoxide (CO)		9.66E+00	4.79E+01	3.63E+01	1.47E+03		6.35E+00	8.39E+00	2.41E+01	9.71E+02	2.58E+03
(a) Methane (CH4)		3.28E+03	1.95E+02	1.08E+02	0.00E+00		2.16E+03	1.13E+02	7.18E+01	0.00E+00	5.93E+03
(a) Nitrogen Oxides (NOx as NO2)	3.21E+03	5.41E+02	8.98E+00	1.40E+02	3.15E+03	1.95E+03	3.56E+02	2.13E+02	9.28E+01	2.09E+03	1.18E+04
(a) Nitrous Oxide (N2O)	2.27E+03	0.00E+00	8.98E+00	4.57E+00	6.70E+00	1.39E+03	0.00E+00	5.09E+00	3.01E+00	4.46E+00	3.70E+03
(a) Sulfur Oxides (SOx as SO2)		1.93E+02	1.90E+03	4.16E+02	4.64E+02		1.27E+02	9.20E+02	2.76E+02	2.61E+01	4.32E+03
(w) Acids (H+)		9.66E-02	4.73E+03	1.04E-01	0.00E+00		6.35E-02	2.22E+03	6.90E-02	0.00E+00	6.95E+03
(w) TN	6.69E+04	1.93E-01	2.73E-01	1.60E+01	0.00E+00	4.39E+04	1.27E-01	1.68E-01	1.05E+01	0.00E+00	1.11E+05
(w) TP	1.84E+03	0.00E+00	5.65E-01	1.94E-02	0.00E+00	1.17E+03	0.00E+00	2.66E-01	1.24E-02	0.00E+00	3.01E+03
(r) Land (ha)	1.91E+06					1.32E+06					3.23E+06
(r) Coal (in ground)		4.62E+03	2.53E+04	4.23E+03			3.04E+03	1.42E+04	2.78E+03		5.42E+04
(r) Natural Gas (in ground)		1.47E+05	4.32E+04	1.16E+04			9.64E+04	2.09E+04	7.61E+03		3.26E+05
(r) Oil (in ground)		6.01E+02	1.11E+04	9.29E+04			3.95E+02	5.56E+03	6.19E+04		1.72E+05
(r) Potassium Chloride (KCl, as K2O, in ground)			6.37E+04					4.80E+04			1.12E+05
(r) Phosphate Rock (in ground)			2.31E+05					1.09E+05			3.40E+05
Water Used (total) (m^3)		2.12E+06	2.67E+05	4.49E+05			1.40E+06	1.28E+05	2.93E+05		4.66E+06

% of total	Corn Farm	Corn FN prod	Corn FP+FK prod	Corn energy prod	Corn energy farm	Soy Farm	Soy FN prod	Soy FP+FK prod	Soy energy prod	Soy energy farm
(a) Ammonia (NH3-N)	70.7	0.0	0.0	0.0	0.0	29.3	0.0	0.0	0.0	0.0
(a) Carbon Dioxide (CO2, fossil)		26.1	6.5	3.2	25.0		17.1	3.6	2.1	16.4
(a) Carbon Monoxide (CO)		0.4	1.9	1.4	57.1		0.2	0.3	0.9	37.7
(a) Methane (CH4)		55.4	3.3	1.8	0.0		36.4	1.9	1.2	0.0
(a) Nitrogen Oxides (NOx as NO2)	27.4	4.6	0.1	1.2	26.8	16.6	3.0	1.8	0.8	17.8
(a) Nitrous Oxide (N2O)	61.4	0.0	0.2	0.1	0.2	37.7	0.0	0.1	0.1	0.1
(a) Sulfur Oxides (SOx as SO2)		4.5	44.0	9.6	10.7		2.9	21.3	6.4	0.6
(w) Acids (H+)		0.0	68.0	0.0	0.0		0.0	32.0	0.0	0.0
(w) TN	60.4	0.0	0.0	0.0	0.0	39.6	0.0	0.0	0.0	0.0
(w) TP	61.2	0.0	0.0	0.0	0.0	38.8	0.0	0.0	0.0	0.0
(r) Land (ha)	59.1	0.0	0.0	0.0		40.9	0.0	0.0	0.0	
(r) Coal (in ground)		8.5	46.8	7.8			5.6	26.2	5.1	
(r) Natural Gas (in ground)		44.9	13.2	3.5			29.6	6.4	2.3	
(r) Oil (in ground)		0.3	6.5	53.9			0.2	3.2	35.9	
(r) Potassium Chloride (KCl, as K2O, in ground)			57.0					43.0		
(r) Phosphate Rock (in ground)			68.0					32.0		
Water Used (total) (m^3)		45.6	5.7	9.6			30.0	2.7	6.3	

Table 29. Overall Annual Cradle-to-Farm Gate LCI Flows and Distributions among Processes, Scenario 2: C-S with Stover Collection

(Scenario 2: C-S w/ stover, allocation method 1)

Cradle to Farmgate Lifecycle Flows (in mt unless noted)	Corn+Stover Farm	Corn+Stover FN prod	Corn+Stover FP+FK prod	Corn+Stover energy prod	Corn+Stover energy farm	Soy Farm	Soy FN prod	Soy FP+FK prod	Soy energy prod	Soy energy farm	TOTAL
(a) Ammonia (NH3-N)	6.47E+04	2.93E-01	3.16E-02	2.84E-01	0.00E+00	2.64E+04	4.22E-03	7.46E-03	2.50E-03	0.00E+00	9.11E+04
(a) Carbon Dioxide (CO2, fossil)		7.47E+05	2.24E+05	4.08E+04	2.88E+05		1.08E+04	5.54E+04	7.02E+03	1.08E+05	1.48E+06
(a) Carbon Monoxide (CO)		1.78E+01	1.24E+02	2.88E+01	1.21E+03		2.56E-01	2.75E+01	3.45E+00	3.13E+02	1.73E+03
(a) Methane (CH4)		6.04E+03	5.12E+02	8.63E+01	0.00E+00		8.70E+01	1.13E+02	1.30E+01	0.00E+00	6.85E+03
(a) Nitrogen Oxides (NOx as NO2)	2.93E+03	9.95E+02	2.22E+01	1.18E+02	2.42E+03	1.15E+03	1.43E+01	2.13E+02	2.47E+01	9.34E+01	7.99E+03
(a) Nitrous Oxide (N2O)	2.68E+03	0.00E+00	2.22E+01	3.84E+00	5.15E+00	6.07E+02	0.00E+00	5.09E+00	8.91E-01	1.86E-01	3.32E+03
(a) Sulfur Oxides (SOx as SO2)		3.55E+02	2.37E+03	3.34E+02	3.66E+02		5.12E+00	9.20E+02	3.10E+01	3.34E+01	4.41E+03
(w) Acids (H+)		1.78E-01	4.88E+03	1.06E-01	0.00E+00		2.56E-03	2.22E+03	2.74E-08	0.00E+00	7.10E+03
(w) TN	1.23E+05	3.55E-01	8.66E-01	1.40E+01	0.00E+00	1.78E+03	5.12E-03	1.68E-01	6.24E+00	0.00E+00	1.25E+05
(w) TP	3.75E+03	0.00E+00	5.85E-01	1.94E-02	0.00E+00	1.17E+03	0.00E+00	2.66E-01	1.59E-02	0.00E+00	4.92E+03
										0.00E+00	
(r) Land (ha)	1.91E+06					1.32E+06					3.23E+06
(r) Coal (in ground)		8.51E+03	6.00E+04	3.59E+03			1.23E+02	1.42E+04	2.08E+03		8.85E+04
(r) Natural Gas (in ground)		2.70E+05	5.37E+04	9.82E+03			3.89E+03	2.09E+04	5.68E+03		3.64E+05
(r) Oil (in ground)		1.11E+03	1.64E+04	7.25E+04			1.59E+01	5.56E+03	3.94E+04		1.35E+05
(r) Potassium Chloride (KCl, as K2O, in ground)			3.35E+05					4.80E+04			3.83E+05
(r) Phosphate Rock (in ground)			2.39E+05					1.09E+05			3.47E+05
Water Used (total) (m^3)		3.91E+06	3.10E+05	4.50E+05			5.63E+04	1.28E+05	2.90E+05		5.15E+06

% of total	Corn+Stover Farm	Corn+Stover FN prod	Corn+Stover FP+FK prod	Corn+Stover energy prod	Corn+Stover energy farm	Soy Farm	Soy FN prod	Soy FP+FK prod	Soy energy prod	Soy energy farm
(a) Ammonia (NH3-N)	71.0	0.0	0.0	0.0	0.0	29.0	0.0	0.0	0.0	0.0
(a) Carbon Dioxide (CO2, fossil)		50.4	15.1	2.8	19.4		0.7	3.7	0.5	7.3
(a) Carbon Monoxide (CO)		1.0	7.2	1.7	70.2		0.0	1.6	0.2	18.1
(a) Methane (CH4)		88.2	7.5	1.3	0.0		1.3	1.6	0.2	0.0
(a) Nitrogen Oxides (NOx as NO2)	36.7	12.5	0.3	1.5	30.3	14.4	0.2	2.7	0.3	1.2
(a) Nitrous Oxide (N2O)	80.6	0.0	0.7	0.1	0.2	18.3	0.0	0.2	0.0	0.0
(a) Sulfur Oxides (SOx as SO2)		8.1	53.6	7.6	8.3		0.1	20.9	0.7	0.8
(w) Acids (H+)		0.0	68.7	0.0	0.0	0.0	0.0	31.3	0.0	0.0
(w) TN	98.6	0.0	0.0	0.0	0.0	1.4	0.0	0.0	0.0	0.0
(w) TP	76.2	0.0	0.0	0.0	0.0	23.7	0.0	0.0	0.0	0.0
(r) Land (ha)	59.1	0.0	0.0	0.0		40.9	0.0	0.0	0.0	
(r) Coal (in ground)		9.6	67.8	4.1			0.1	16.0	2.3	
(r) Natural Gas (in ground)		74.2	14.8	2.7			1.1	5.7	1.6	
(r) Oil (in ground)		0.8	12.2	53.7			0.0	4.1	29.2	
(r) Potassium Chloride (KCl, as K2O, in ground)			87.4					12.6		
(r) Phosphate Rock (in ground)			68.7					31.3		
Water Used (total) (m^3)		76.0	6.0	8.8			1.1	2.5	5.6	

Table 30. Overall Annual Cradle-to-Farm Gate LCI Flows and Distributions among Processes, Scenario 3: C-C with Stover Collection

(Scenario 3: C-C w/ stover)

Cradle to Farmgate Lifecycle Flows (in mt unless noted)	Corn+Stover Farm	Corn+Stover FN prod	Corn+Stover FP+FK prod	Corn+Stover energy prod	Corn+Stover energy farm	TOTAL
(a) Ammonia (NH3-N)	1.15E+05	6.62E-01	4.10E-02	4.94E-01	0.00E+00	1.15E+05
(a) Carbon Dioxide (CO2)		1.69E+06	2.95E+05	7.11E+04	5.71E+05	2.63E+06
(a) Carbon Monoxide (CO)		4.02E+01	1.58E+02	5.02E+01	2.11E+03	2.36E+03
(a) Methane (CH4)		1.37E+04	6.49E+02	1.50E+02	0.00E+00	1.45E+04
(a) Nitrogen Oxides (NOx as NO2)	6.60E+03	2.25E+03	2.85E+01	2.06E+02	4.22E+03	1.33E+04
(a) Nitrous Oxide (N2O)	6.25E+03	0.00E+00	2.85E+01	6.69E+00	8.98E+00	6.29E+03
(a) Sulfur Oxides (SOx as SO2)		8.04E+02	3.72E+03	5.83E+02	6.38E+02	5.75E+03
(w) Acids (H+)		4.02E-01	8.27E+03	1.84E-01	0.00E+00	8.27E+03
(w) TN	2.82E+05	8.04E-01	1.06E+00	2.44E+01	0.00E+00	2.82E+05
(w) TP	5.71E+03	0.00E+00	9.90E-01	3.38E-02	0.00E+00	5.71E+03
(r) land (ha)	3.23E+06					3.23E+06
(r) Coal (in ground)		1.92E+04	7.78E+04	6.26E+03		1.03E+05
(r) Natural Gas (in ground)		6.10E+05	8.46E+04	1.71E+04		7.12E+05
(r) Oil (in ground)		2.50E+03	2.43E+04	1.26E+05		1.53E+05
(r) Potassium Chloride (KCl, as K2O, in ground)			3.77E+05			3.77E+05
(r) Phosphate Rock (in ground)			4.05E+05			4.05E+05
Water Used (total) (m^3)		8.84E+06	5.02E+05	7.84E+05		1.01E+07

% of total	Corn+Stover Farm	Corn+Stover FN prod	Corn+Stover FP+FK prod	Corn+Stover energy prod	Corn+Stover energy farm
(a) Ammonia (NH3-N)	100.0	0.0	0.0	0.0	0.0
(a) Carbon Dioxide (CO2, fossil)	0.0	64.3	11.2	2.7	21.8
(a) Carbon Monoxide (CO)	0.0	1.7	6.7	2.1	89.5
(a) Methane (CH4)	0.0	94.5	4.5	1.0	0.0
(a) Nitrogen Oxides (NOx as NO2)	49.6	16.9	0.2	1.5	31.7
(a) Nitrous Oxide (N2O)	99.3	0.0	0.5	0.1	0.1
(a) Sulfur Oxides (SOx as SO2)	0.0	14.0	64.8	10.1	11.1
(w) Acids (H+)	0.0	0.0	100.0	0.0	0.0
(w) TN	100.0	0.0	0.0	0.0	0.0
(w) TP	100.0	0.0	0.0	0.0	0.0
(r) Land (ha)	100.0				
(r) Coal (in ground)		18.6	75.3	6.1	
(r) Natural Gas (in ground)		85.7	11.9	2.4	
(r) Oil (in ground)		1.6	15.9	82.5	
(r) Potassium Chloride (KCl, as K2O, in ground)		0.0	100.0	0.0	
(r) Phosphate Rock (in ground)		0.0	100.0	0.0	
Water Used (total) (m^3)		87.3	5.0	7.7	

Table 31. Summary of Changes in Flows and Incremental Percent Differences in LCI Flows Among Scenarios[a]

Scenario 2: C-S with stover collection (allocation method 1)

Cradle to Farmgate Lifecycle Flows (in mt unless noted)	Corn+Stover Farm	Corn+Stover FN prod	Corn+Stover FP+FK prod	Corn+Stover energy prod	Corn+Stover energy farm	Soy Farm	Soy FN prod	Soy FP+FK prod	Soy energy prod	Soy energy farm	TOTAL change	% change
(a) Ammonia (NH3-N)	9.98E+02	3.33E-02	1.83E-02	5.17E-03	0.00E+00	0	0	0	0	0	9.98E+02	1.1
(a) Carbon Dioxide (CO2, fossil)		8.48E+04	1.24E+05	-9.19E+03	-1.01E+05		0	0	-2.61E+04	-1.48E+05	-7.56E+04	-5.1
(a) Carbon Monoxide (CO)		2.02E+00	7.61E+01	-7.53E+00	-2.60E+02		0	0	-2.07E+01	-6.58E+02	-8.68E+02	-50.3
(a) Methane (CH4)		6.87E+02	3.17E+02	-2.20E+01	0.00E+00		0	0	-5.88E+01	0.00E+00	9.23E+02	13.5
(a) Nitrogen Oxides (NOx as NO2)	-1.07E+03	1.13E+02	1.33E+01	-2.22E+01	-7.24E+02	0	0	0	-6.82E+01	-2.00E+03	-3.76E+03	-47.1
(a) Nitrous Oxide (N2O)	-3.84E+02	0.00E+00	1.33E+01	-7.28E-01	-1.55E+00	0	0	0	-2.12E+00	-4.27E+00	-3.79E+02	-11.4
(a) Sulfur Oxides (SOx as SO2)		4.04E+01	4.67E+02	-8.12E+01	-9.77E+01		0	0	-2.45E+02	7.37E+00	9.06E+01	2.1
(w) Acids (H+)		2.02E-02	1.47E+02	1.97E-03	0	0	0	0.00E+00	-6.90E-02	0	1.47E+02	2.1
(w) TN	1.40E+04	4.04E-02	5.93E-01	-2.01E+00	0	0	0	0.00E+00	-4.23E+00	0	1.40E+04	11.2
(w) TP	1.91E+03	0.00E+00	2.03E-02	2.46E-07	0	0	0	0.00E+00	0	0	1.91E+03	38.8
(r) Land (ha)	0					0					0.00E+00	0.0
(r) Coal (in ground)		9.66E+02	3.47E+04	-6.39E+02			0	0	-7.01E+02		3.43E+04	38.7
(r) Natural Gas (in ground)		3.06E+04	1.05E+04	-1.76E+03			0	0	-1.93E+03		3.75E+04	10.3
(r) Oil (in ground)		1.26E+02	5.28E+03	-2.04E+04			0	0	-2.25E+04		-3.75E+04	-27.8
(r) Potassium Chloride (KCl, as K2O, in ground)			2.71E+05					0			2.71E+05	70.8
(r) Phosphate Rock (in ground)			7.18E+03					0			7.18E+03	2.1
Water Used (total) (m^3)		4.44E+05	4.30E+04	1.65E+03			0	0	-3.13E+03		4.86E+05	9.4

Scenario 3: C-C w/ stover collection

Cradle to Farmgate Lifecycle Flows (in mt unless noted)	Corn+Stover Farm	Corn+Stover FN prod	Corn+Stover FP+FK prod	Corn+Stover energy prod	Corn+Stover energy farm	TOTAL change	% change
(a) Ammonia (NH3-N)	5.15E+04	4.02E-01	2.77E-02	2.15E-01	0.00E+00	2.51E+04	27.5
(a) Carbon Dioxide (CO2, fossil)		1.03E+06	1.95E+05	2.11E+04	1.82E+05	1.07E+06	72.2
(a) Carbon Monoxide (CO)		2.44E+01	1.10E+02	1.39E+01	6.39E+02	-2.35E+02	-13.6
(a) Methane (CH4)		8.31E+03	4.54E+02	4.22E+01	0.00E+00	8.53E+03	124.5
(a) Nitrogen Oxides (NOx as NO2)	2.60E+03	1.37E+03	1.96E+01	6.56E+01	1.08E+03	1.56E+03	19.5
(a) Nitrous Oxide (N2O)	3.19E+03	0.00E+00	1.96E+01	2.12E+00	2.28E+00	2.59E+03	78.0
(a) Sulfur Oxides (SOx as SO2)		4.89E+02	1.82E+03	1.67E+02	1.74E+02	1.43E+03	32.3
(w) Acids (H+)		2.44E-01	3.54E+03	8.00E-02	0	1.32E+03	18.5
(w) TN	1.73E+05	4.89E-01	7.83E-01	8.39E+00	0	1.71E+05	136.9
(w) TP	3.87E+03	0.00E+00	4.25E-01	1.44E-02	0	2.70E+03	54.9
(r) Land (ha)	1.32E+06					0.00E+00	0.0
(r) Coal (in ground)		1.17E+04	5.25E+04	2.03E+03		4.91E+04	55.5
(r) Natural Gas (in ground)		3.71E+05	4.14E+04	5.54E+03		3.85E+05	105.9
(r) Oil (in ground)		1.52E+03	1.32E+04	3.35E+04		-1.92E+04	-14.3
(r) Potassium Chloride (KCl, as K2O, in ground)			3.14E+05			2.66E+05	69.5
(r) Phosphate Rock (in ground)			1.73E+05			6.43E+04	18.5
Water Used (total) (m^3)		5.38E+06	2.34E+05	3.35E+05		5.47E+06	106.3

[a] change and percent increase relative to base case scenario

5. Results – Lifecycle Impact Assessment

The environmental implications for C-S agriculture and stover harvesting were evaluated through assessment of three specific environmental impacts: eutrophication, acidification and green house gas generation. The impact of nitrate as a toxic constituent in groundwater was also evaluated, although it is recognized that this analysis integrates only one of many potential toxic components that would be released from the cradle-to-farm gate stages of the C-S or C-C lifecycles.

Eutrophication Potential

The relationship between fertilizer use in row crop agriculture and local and regional scale eutrophication problems has been recognized for decades (USEPA, 2000a). Both nitrogen an phosphorus contribute as nutrients used in the growth of excess aquatic biomass, although in certain cases, one or the other is the truly limiting nutrient.

In life cycle assessments, equivalency factors are used to aggregate nitrogen and phosphorus species to quantify the overall eutrophication potential. The molar ratio (16:1) for the N:P content in aquatic cell mass is used to quantify the equivalency factors (Hauschild and Wenzel, 1998). If the higher N:P ratio of 22:1 for eutrophic waters is used (Klaff, 2003), results would show greater contributions from TP and higher total eutrophication potentials with this higher ratio.

The eutrophication potential calculated here includes only the (w) TN-SW and (w) TP LCI flows presented in Tables 21-23. None of the other nitrogen or phosphorus flows are significant relative to these releases. The eutrophication potential (mt as NO_3-N equivalents) discharged to local streams and then the Mississippi River is compared to several different benchmarks to define the potential impact of these discharges. Bench marks include:

- Background concentration of nitrate in streams and rivers (Table 2)
- Recommended water quality standards for TN and TP (Table 2) (as total equivalent NO_3-N)
- Recommended 30% reduced load of 1980-1996 average TN load to the Gulf of Mexico (NOAA Integrated Assessment of Hypoxia in the Gulf of Mexico, Brezonik et al., 1999)

Concentration limits were multiplied by the actual annual mean stream flow rate and converted to a eutrophication potential (TN+TP, as NO_3-N) to determine a load. It was assumed that the eastern Iowa watersheds would reduce their average TN loads at the same percentage as the entire Mississippi River watershed. A target load that would limit the hypoxic zone to an acceptable size was determined as a 30% reduction in the average TN load from E. Iowa (1980-1996) for comparison purposes.

The calculated eutrophication potential for the 13 years of the study period is shown on Figure 34. The base case scenario is represented by the solid region of the bars, with the incremental increases associated with a switch to scenario 2 or 3 included as the diagonal hashed region of each bar. Many of the conclusions that can be drawn from this figure parallel those already discussed for individual LCI flows.

- The eutrophication potential varies substantially by year, due primarily due to the increased rainfall and means of transporting nitrogen and phosphorus from the field to surface water.
- TN loads contribute more to the total eutrophication potential than TP loads.
- A switch to scenario 3 more than doubles the total eutrophication potential (~150% increase over the base case). Both the increased FN required due to the loss of the soy credit and the assumed higher TP leaching rate in the NT system contribute to this significant rise in the eutrophication potential.

- Increases in the eutrophication potential for scenario 2 are less substantial (~20% increase over the base case) than in scenario 3.

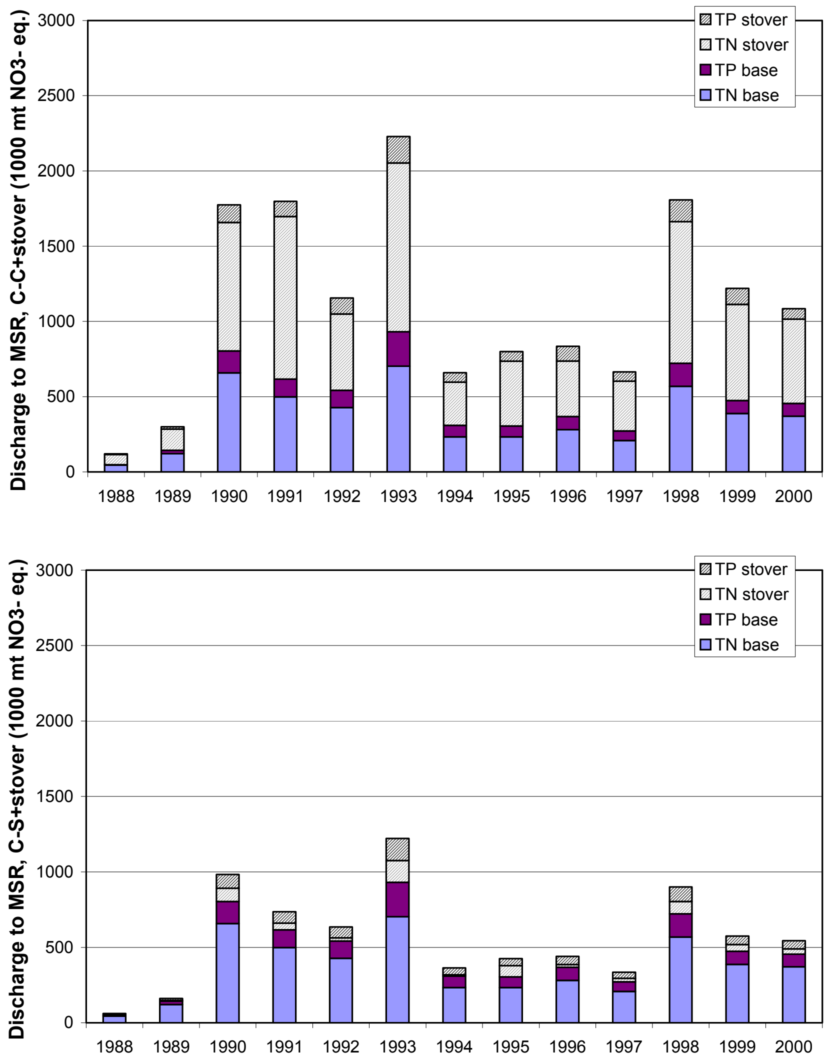


Figure 34. Eutrophication potential from TN and TP discharged from E. IA watersheds to the Mississippi River. Base case included in each as solid regions. Hash marks include the incremental addition associated with scenario 2 (bottom) and scenario 3 (top)

The eutrophication potential for the base case is compared to the applicable benchmarks in Figure 35. The estimated discharge, expressed as an eutrophication potential, is much higher in all years than the load at a concentration equal to the recommended water quality standards. This indicates that unacceptably high levels of biomass growth, decay, and oxygen depletion will occur in the streams and rivers in eastern Iowa. As summarized in Figure 1, habitat destruction, loss of biodiversity, and an increase in water toxics all result from eutrophication.

As the TN and TP are transported downstream in the Mississippi River, they have additional eutrophication consequences in the Gulf of Mexico where they contribute to hypoxia. The goal of a 30% reduction in the average TN load established by a federally endorsed task force on hypoxia (Brezonik et al., 1999) is shown as a straight line at ~500,000 mt as NO_3-N.

The E. IA contributions to the total eutrophication potential exceed this goal in 5 of the 13 years in this study. Modeling work by Scavia et al. (2003) suggest that the goal set by Brezonik et al. (1999) is too low and that a 40-45% reduction in average TN loads would be required to limit the size of the hypoxic zone. The 45% reduction in average TN loads would lower the acceptable eutrophication potential to 421,000 mt as NO_3-N. TN and TP contributions from C-S agriculture in E.IA would exceed this goal in 7 of the 13 years.

In contrast, the use of a constant fraction of FN (24%) to estimate the nitrate load to surface water, such as used in other LCAs, with average TP loads would predict that the eutrophication potential discharged from E. Iowa to the MSR is always less than the suggested ~500,000 mt NO_3-N target required for a 30% load reduction to reduce hypoxia. This example clearly shows the necessity of incorporating the climatic variability in loads on actual environmental impact.

It should be noted that the discussion above addresses only one source of TN and TP discharges within the eastern Iowa watersheds. Animal manure can also contribute additional nutrients to water bodies. Based on the work Becher et al. (2000), animal manures can contribute an additional 34% above the loads from FN. Adding this to the base case scenario shown in Figure 35 would cause the 1994-1997 loads to also exceed the TN load target of a 45% reduction.

Increasing the nutrient loads to the Mississippi River and the Gulf of Mexico under scenarios 2 or 3 will move us further away from meeting necessary levels to limit local eutrophication or hypoxia conditions. For the C-S scenario with stover removal (#2), the eutrophication potential would be ~20% higher than the base case. The total load (including animal manure) would be above 421,000 mt as NO_3-N (45% reduction goal) in all years except extreme drought. The very substantial increases in eutrophication potential loads associated with scenario 3 would certainly increase these loads above acceptable levels (30 or 45% reductions in TN) in all years except 1988 and 1989.

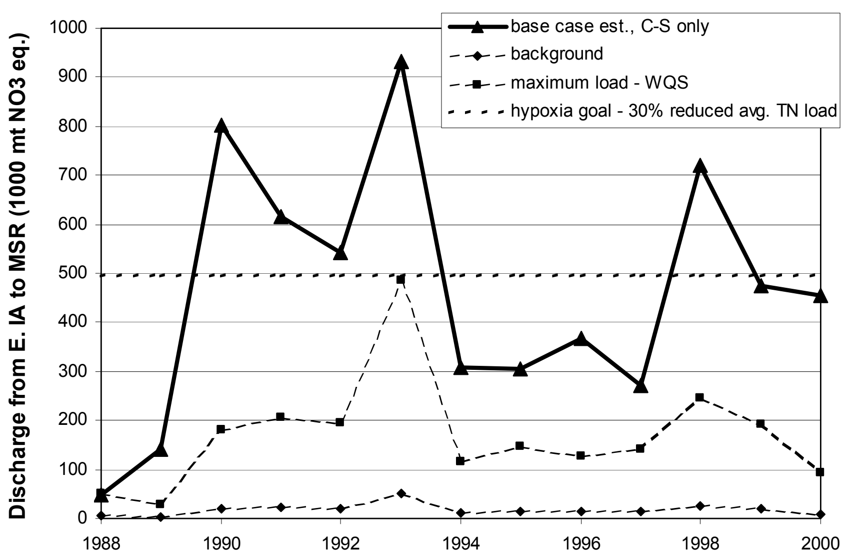


Figure 35. Base case eutrophication potential discharged to the Mississippi River compared with suitable benchmarks

Figure 36 compares the average discharge of TN and TP, expressed as eutrophication potential for the three scenarios. Any changes in the present system that result in increased fertilizer use or increased leaching will only exacerbate local eutrophication and hypoxia problems. Thus, any changes in agricultural practices that would allow stover collection should be designed to minimize greater levels of leaching. For the C-C scenario, the much higher discharge of TN and TP to a system that has already exceeded its capacity to assimilate nutrients makes it difficult to recommend this option for increased stover harvest rates.

The much lower TN and TP discharges from C-S rotations with stover harvest makes this scenario much more attractive than scenario 3. Careful farming practices that limit the fertilizer addition to just that amount deemed necessary from quantitative soil analysis would probably be sufficient to further reduce the nutrient loads to surface water bodies to levels acceptable from a hypoxia standpoint in most years. It is likely that recommended water quality standards designed to limit local eutrophication problems would still be exceeded, however.

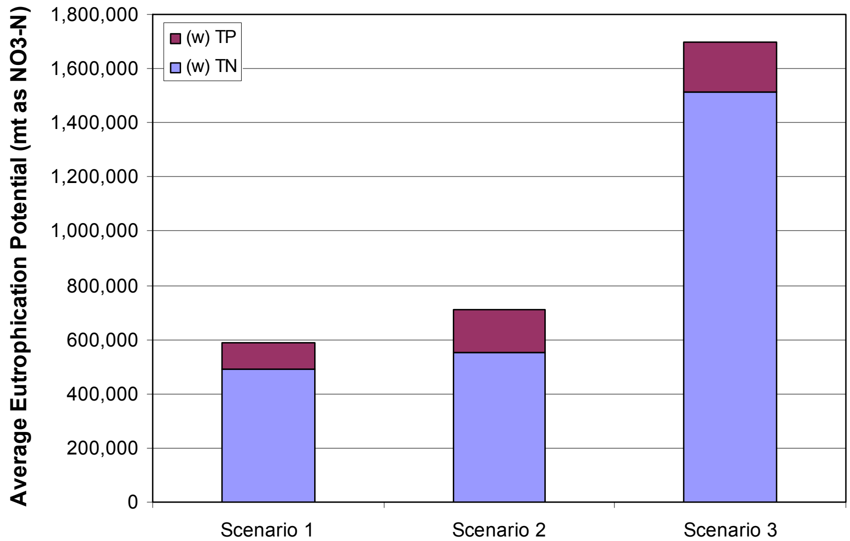


Figure 36. Average eutrophication potentials for the three scenarios

The analysis presented here is limited by several assumptions. Most importantly,

- It was assumed that 45 kg/ha additional FN would be required for C-C rotations due to the loss of the symbiotic relationship with soy.
- TP leaching rates were 1.6 times higher in the no till system
- TN and TP discharges were assumed to not be correlated to erosion rates, which are substantially lower with no till practices.

Each of these assumptions contributes to high-end estimates for incremental increases in nutrient discharges for the stover collection scenarios. None of these assumptions, however, would impact the predicted high nutrient discharges associated with the base case. The quality of predictions for the base case scenario is impacted most by the nitrate and phosphorus leaching models. Uncertainties in how animal manure and denitrification were included in these models could impact the conclusions drawn here. Additional work is required to quantify these uncertainties.

Acidification Potential

In this cradle-to-farm gate LCA, air emissions that contribute to acidification include NH_3 and NO from nutrient use and plant growth at the farm and NO_x and SO_x from fertilizer and energy production. HCl and HF emissions were also quantified, but they were insignificant. Air emissions contributing to acidification were aggregated into an overall acidification potential for each of the scenarios using the equivalency factors presented by Hauschild and Wenzel (1998). Average values over the 13-year period were used to quantify air emissions associated with the farming activities.

The overall acidification potential for scenario 1 is shown in Figure 37. Ammonia releases from the farm are the predominant contributor to the overall acidification potential. Volatilization of FN contributes a small fraction to this release. Plant senescencing is the biggest source of ammonia emissions. As described above, there are significant uncertainties quantifying this source. The values used here are approximately half those used in some studies, although others ignore this term completely. Thus, the range for ammonia could range from near zero to almost twice the emissions shown in Figure 36.

NO released during soil processes of nitrification and denitrification contributes 44% of the total NO_x (as SO_2). Much of this is released in the necessary nitrification of ammonia fertilizers into nitrate. Some additional NO is released during soil mineralization. The 44% estimate here is much lower than Sheehan et al's (2002) estimate that 84% of the total NO_x is due to soil processes. Based on IPCC guidelines, they estimated that 5% of the total nitrogen in FN is emitted as NO. The NO emissions estimated with the model used here, which more directly relates NO to actual nitrification and denitrification processes, were determined to be ~2% of the FN (Table 19).

Other sources of NO_x and SO_x are mostly related to the combustion of fossil fuels for energy production and farming activities. The apparent contributions of FK+FP production to these emissions are dominated by fossil fuel combustion for the required electricity.

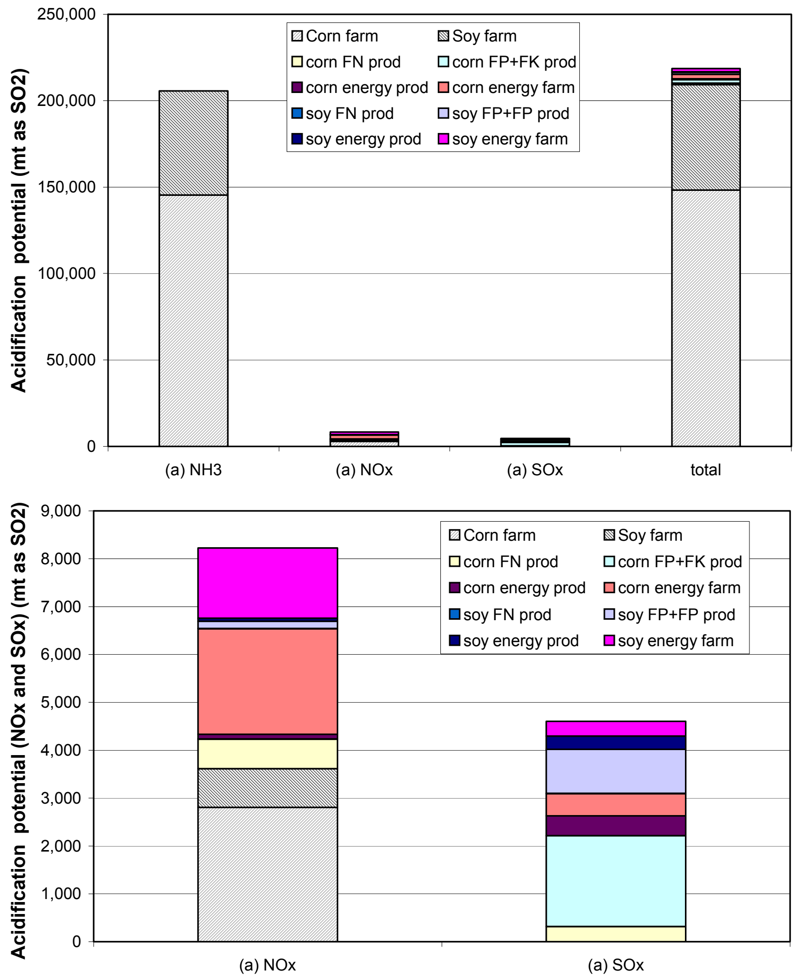


Figure 37. Cradle-to-farm gate acidification potential for Scenario 1: C-S with no stover harvest base case. The bottom figure provides a detailed view of the same NO_x and SO_x data included in the top figure.

Figure 38 provides a comparison in the average acidification potential for the three scenarios. There is less difference in acidification among the scenarios compared with eutrophication. This is due to the very high contribution of plant senescencing to the total ammonia emissions. This process occurs regardless of the farming practices. The higher acidification potential for scenario 3 is due to the higher senescing rate from corn versus soy. The slightly lower NO_x contribution in scenario 2 is due to the reduced soil mineralization – and associated NO emissions – with the no till practice. This benefit does not contribute substantially to the overall acidification potential.

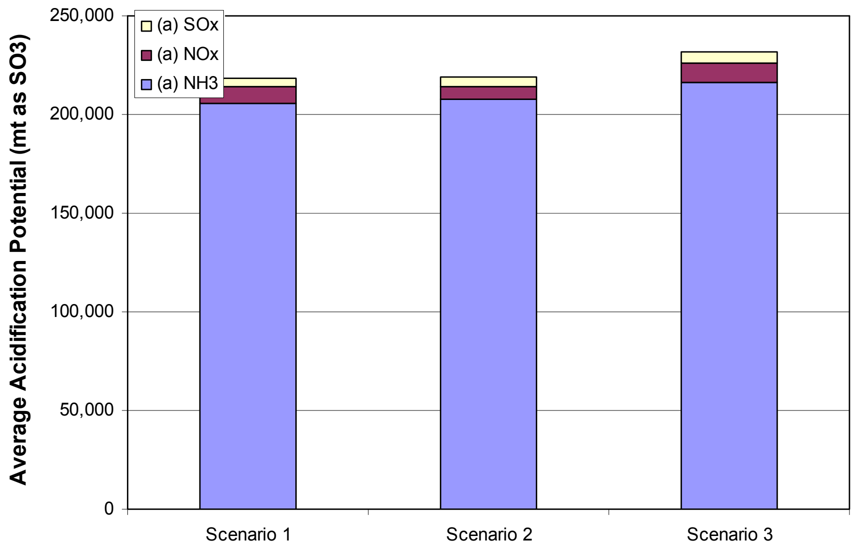


Figure 38. Average acidification potentials for the three scenarios

Water Toxics

Nitrate is the only toxic species considered in the analysis conducted here. Herbicides and slightly soluble species from fuel spills at the farm are important toxic constituents that should also be considered in a more comprehensive analysis of this impact category.

The MCL for nitrate-nitrogen (10 mg/L) can be considered a reasonable benchmark for defining the impact of nitrate on water quality from a toxic perspective. This impact is especially important for ground water ingestion due to the high percentage of people in rural areas that use ground water as drinking water.

The model used for the LCI predicts that 4,420 mt of NO_3-N are leached into the groundwater over the 13 year period in the base case. Although concentration data were not predicted in this model, the total NO_3-N load can be used to estimate the volume of water contaminated by the leaching of FN. Assuming water is contaminated at the maximum allowable concentration, the nitrate could pollute 4.4×10^8 m^3 water. This assumes that none of the nitrate is denitrified and that groundwater flows are slow relative to the 13-year period of this study. Assuming the elevated nitrate concentrations remain in the top 2 m of the saturated zone and an average porosity of 0.25, this water could include an area of 8.8×10^4 ha. This represents 3.1% of the total C-S farmland. This is a low estimate compared with the 9 - 11.6% of the shallow groundwater wells in E. Iowa that Kross et al. (1993) observed to have concentrations greater than the MCL. Uncertainty in this analysis includes the leaching rate (3% of excess nitrate; eqn. [26]) and the depth of water contaminated over the 13-year period. Uncertainty in both of these factors could result in greater areas with groundwater contamination at a level greater than acceptable.

This same analysis can be applied to the much higher predicted NO_3-N mass leached to groundwater from the continuous corn scenario (#3). In this case, 121,108 mt NO_3-N are predicted to leach into the groundwater over the 13-year period. The orders of magnitude increase (x27.4) is due to the increased FN requirements for corn versus soybeans. This also includes 45 kg-N/ha additional nitrogen required to replace the symbiotic benefits of the C-S rotation. The total leaching rate would be lower without this additional FN, but still substantially higher than in the base case. With the rough estimation approach applied above, this would result in 1.2×10^{10} m^3 water over an area of 2.4×10^6 ha, which is 84% of the total farmland in the eastern Iowa region that could be estimated to have NO_3-N concentrations greater than the MCL.

Although there are numerous uncertainties in this analysis, it is clear that switching to scenario 3 (C-C) could lead to a substantial rise in exposure of rural populations to unacceptably high nitrate concentrations through their drinking water.

The major uncertainties in this analysis include:

- The required FN rate for C-C, NT operation
- Quantity of nitrate that leaches into the saturated groundwater zone in this region with significant tile drainage
- Extent of denitrification of the nitrate in the groundwater over the 13-year period
- Depth of contaminated groundwater

The application of a numerical model that improves the nitrate leaching estimates (e.g., RZWQM, GLEAMS, AGNPS) could substantially reduce the uncertainties associated with this analysis.

Green House Gas Emissions

Green house gas emissions associated with fossil fuel use and soil processes generating N_2O were calculated. Equivalency factors for 100-year global warming potential (GWP) were used to convert CH_4 and N_2O emissions to CO_2 equivalents (Hauschild and Wenzel, 1998). The total global warming potential for the base case scenario is shown in Figure 39. Carbon dioxide emissions from fossil fuel combustion are the predominant GHG. N_2O from soil processes is also significant, however, contributing 41% of the total GWP.

Data in Figure 39 does not include carbon dioxide removed from the system as biomass or a net increase in the soil carbon. Brenner et al. (2001) used the CENTURY model to estimate increases in carbon sequestered in soil in each Iowa county. These data were used to estimate the carbon sequestered on C-S acreage in each of the counties considered here. On average, 1.1 (±0.5) mt CO_2/ha are sequestered under current conditions. This translates to 3.2 million mt/y CO_2 sequestered in the soil on C-S cropland. Comparing this value to the GHG emissions shown in Figure 40, it can be concluded that the carbon sequestered in the soil is sufficient to balance the GWP associated with fossil fuel energy combustion and soil processes for scenarios 1 and 2.

Assuming an average carbon content of 0.4 kg C/kg dry mass in the corn and soy grain, another 30 million mt/y CO_2 is sequestered in grain that leaves the system as defined here. Much of this is released in a relatively short time after the grain is used for feed or fuel production, so has little impact on the net GWP.

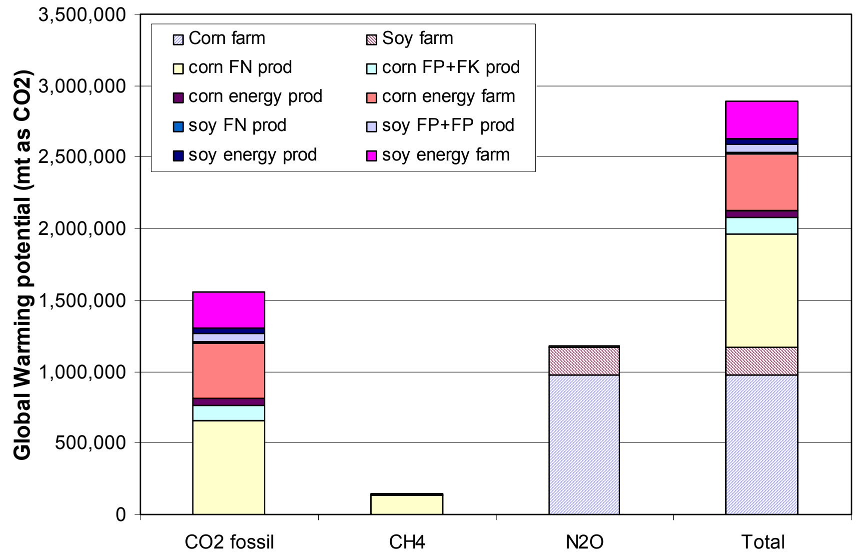


Figure 39. Average global warming potential for the base case scenario

Figure 40 illustrates the average GWP for each of the three scenarios. The substantial increases in GWP noted for scenario 3 are related to the increase in FN use. This increase translates into an increase in energy consumed to manufacture the FN as well as N_2O released in nitrogen transformation processes. The slight reduction in GWP for scenario 2 is due to the assumed reduction in soil mineralization with no till practices. This reduction would also be accompanied by carbon sequestration at rates even higher than estimated by Brenner et al. (2001) for current conditions.

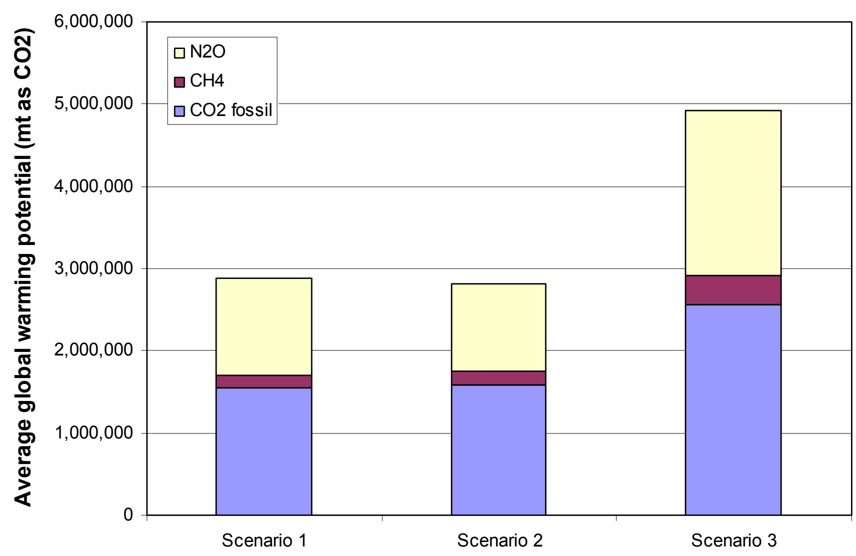


Figure 40. Average global warming potentials for the three scenarios

Summary of Life Cycle Impacts

The section above quantifies changes in eutrophication, global warming and acidification potentials for these three impact categories to provide a clearer picture of the net impacts associated with stover collection. The qualitative discussion of the toxicity of nitrate in groundwater does not provide a complete accounting of the human toxicity potential and is not considered further. Table 32 summarizes the total impacts for each of these three categories as well as the percent change for each of the stover collection scenarios relative to the base case.

Table 32. Summary of Life Cycle Impact Analysis

Impact Category	Scenario 1	Scenario 2	Scenario 3
	Total Loads from E. Iowa (1000 mt)		
Eutrophication Potential (as NO_3-N)	587	711	1,699
Acidification Potential (as SO_2)	219	219	232
Global Warming Potential (as CO_2)	2,886	2,810	4,924
	% increase over base case		
Eutrophication Potential (as NO_3-N)	--	21	189
Acidification Potential (as SO_2)	--	0	6
Global Warming Potential (as CO_2)	--	-3	71
	Normalized Loads[a] (mt/mt stover harvested)		
Eutrophication Potential (as NO_3-N)	--	0.072	0.093
Acidification Potential (as SO_2)	--	0.022	0.013
Global Warming Potential (as CO_2)	--	0.286	0.271

[a] using 13-year average stover harvest yields: ~9.8 million mt for scenario 2 and 18.2 million mt for scenario 3

When compared on a normalized basis (per mt stover harvested), scenario 3 looks slightly better than scenario 2 in terms of acidification and global warming. The eutrophication potential for scenario 3, however, is almost 30% higher than scenario 2.

Of these three categories, eutrophication could be considered the most important because the discharge of nutrients even under the base case already exceeds measures of acceptable discharge levels, whereas the global warming potential is balanced with agricultural benefits of carbon sequestration in soil. Based on eutrophication, it is clear that switching to a continuous corn scenario could be very detrimental to water quality. Increases in local eutrophication and, more importantly, hypoxia in the Gulf of Mexico would be much higher for C-C that C-S, even when normalized to the amount of stover collected.

There are significant uncertainties in the numerical values presented in Table 32 due to the assumptions required in this analysis. The LCI for scenario 1 was calibrated and verified as much as possible with mass balances and historical data for farming and water quality. Several assumptions were required to extrapolate the nutrient fate model to the other two scenarios. For example, the quantitative results from the eutrophication potential in scenario 2 depends on the assumed increased phosphate leaching in a no till scenario and the additional fertilizer requirements due to the removal of corn stover. In addition, estimates for scenario 3 are affected by the assumption that more fertilizer is required for corn in a continuous corn system versus a corn-soybean rotation. The importance of nitrogen cycle symbiosis in a C-S system is known, however, the quantitative amount of extra FN required in a C-C system to replace this symbiosis is not well understood. Processes such as this should be studied in more detail to determine the sensitivity of the conclusions to these uncertainties.

6. Summary and Recommendations

Fertilizers used to increase the yield of crops used for food or bio-based products can migrate through the environment and potentially cause adverse environmental impacts. Nitrogen fertilizers have a complex biogeochemical cycle. Through their transformations and partitioning among environmental compartments, they can contribute to eutrophication of surface waters at local and regional scales, groundwater degradation, acid rain, and global warming. Phosphate fertilizers have a simpler fate in the environment, although leaching of soluble and bound phosphorus is an important contributor to eutrophication.

Eutrophication is considered one of the most pervasive problems affecting water quality in the United States, especially in the Midwest where fertilizers are used extensively for agriculture. In the process of eutrophication, the presence of excess N and P nutrients allows over production of plant biomass in waterways. The eventual degradation of this biomass consumes oxygen resulting in hypoxic conditions (low oxygen concentrations) in the most severe cases of eutrophication. Fertilizer use on corn and soybean farms in the Midwest is considered one of the primary contributors to the growing hypoxic zone in the Gulf of Mexico. Through a combination of excessive nutrient loads and hydrodynamic conditions, a region along the coast of Louisiana that is approximately the size of the State of Massachusetts is considered ecologically dead most summers. This results in the death of species that are not sufficiently mobile and changes in biodiversity and food webs throughout the region as larger species migrate to other locations. Researchers for federal agencies have suggested that reducing the average nitrogen load by 30% will help to limit the hypoxic zone to acceptable levels. Other researchers predict that a 40-45% reduction in TN would be required to meet this goal.

With an increase interest in the use of corn, soybeans and corn stover for bio-based products and fuels, it is important to understand the relative environmental benefits and deleterious impacts associated with this growing market. A team of researchers lead by NREL completed a life cycle assessment (LCA) for stover harvest and conversion to ethanol for transportation fuels. This report focused on the green house gas emission benefits associated with biofuels as balanced by potential detriments to soil health (carbon content and erosion). Eutrophication was identified as an important issue by stakeholders involved with this project, but resources prevented this environmental impact category from being addressed. Thus, the goal of the work presented here was primarily to fill that gap, thereby providing a more complete picture of the overall environmental impacts associated with bio-based products.

The mathematical model developed here to track the flows of nutrients through the environment and harvested crops quantifies the distribution of nutrients as a fraction of the available N or P. Knowledge of specific processes was integrated into this model as much as possible to correlate emissions to the specific processes that affect them. For example, N and P leaching are correlated to rainfall and NO and N_2O emissions vary between conventional and no till practices due to reduced soil mineralization.

The nutrient flow model was applied to a set of watersheds covering most of the eastern half of Iowa for a 13-year period (1988 – 2000) that includes both drought and flood years. Water quality data were available for these watersheds allowing the TN and TP leaching estimates to be calibrated to actual measured discharges. This approach has not been used before with LCAs for non-point source pollutants and provides a much better approach for quantifying the variability in leaching rates than approaches used by other researchers. The resulting leaching model incorporates variability in TN and TP loads due to the natural variability in annual rainfall. Although this approach helps to improve the estimates and variability in non-point emissions from nutrient leaching, it also makes the model and approach very site specific. Thus, the specific linear model presented here to estimate the fraction of fertilizer that leaches should not be applied to other locations. For example, western Iowa has sandier soil, which would result in greater quantities of nutrients to infiltrate to groundwater and less surface runoff. Although the specific values cannot be used for other sites, the general approach and framework is valuable and can be used at other sites that have suitable water quality data.

The nutrient leaching model was coupled with LCA data describing emission occurring during fertilizer manufacture and energy production and consumption, providing a cradle-to-farm gate life cycle inventory for corn, soybeans and stover. Three separate scenarios were considered: Scenario 1, the base case, considered corn-soybean rotations (C-S) with conventional till and no stover collection; Scenario 2 was C-S with no till and stover collection at a maximum rate allowable with acceptable erosion levels; and, Scenario 3 was the same as 2 except for continuous corn (C-C) rather than C-S. The LCI for each of these scenarios was quantified and used to determine eutrophication, acidification and global warming potentials for the three scenarios.

The TN and TP leaching models that correlate the fraction of FN and FP that leaches to annual rainfall in each county provides a good representation of the measured variability in nutrient loads discharged to the Mississippi River. The ability to calibrate this model reduces the uncertainty in the leaching estimates for the base case scenario than would be possible without the site-specific water quality data. There are, however, more significant uncertainties associated with the allocation of the total load between corn and soy and the predictions of loads when moving to the no till, stover collection scenarios. Two radically different approaches were used to estimate the allocation of nitrogen flows between corn and soybeans. For example, between the two methods, between 60 and 99% of the TN leached from the overall C-S system would be allocated to corn, with the balance allocated to soybeans. This difference stems from a poorly understood symbiosis with the C-S rotation. It is not apparent how to best integrate this into an LCA.

Analysis of the LCI data identified several variables and processes for which the model is very sensitive. These include:

- Allocation of nitrogen flows between corn and soybeans
- Ammonia released during plant senescencing
- TN and TP leaching models, especially contributions of manure to total load to the MSRB
- Fertilizer use and fate in a NT system versus convention till
- Fertilizer requirements for a C-C versus C-S system
- Actual erosion rates
- Soil mineralization rates in a no till versus conventional till system.

Further work to improve the values of the parameters is required, as well as quantification of the sensitivity of the conclusions drawn here to uncertainties in these processes. The overall conclusions presented below do not integrate any sensitivity analysis at this point.

The primary focus of this research was the eutrophication potential associated with the three scenarios. TN and TP flows to the Mississippi River and Gulf of Mexico were aggregated into an overall eutrophication potential, expressed as equivalent mass of NO_3-N. The eutrophication loads predicted through the nutrient flow model developed here were compared to the target reduction in TN flows to limit the size of the Gulf of Mexico hypoxic zone and recommended water quality standards to control eutrophication at a local level.

The results of this analysis show that the eutrophication potential for the base case already exceeds acceptable limits. TN and TP discharges from C-S lands exceed the maximum loads defined by the proposed water quality standards each of the 13 years of this study. Limits established by the goal of a 30% reduction in the average TN load are also exceeded in approximately half of the years. If additional sources of nutrients discharged from the study area are considered, it is estimated that the eutrophication potential is higher than recommended except under drought conditions.

Changing the current conditions to harvest stover for biofuels production increases the eutrophication potential above the base case. The C-S scenario (2) results in a 21% increase in the eutrophication potential, while the C-C scenario (3) causes almost triple the total load of TN and TP (as equivalent NO_3-N). In either case, the increased in load is to a system that has already exceeded its assimilative capacity for nutrients. For the C-S system, it is likely that careful management of fertilizers used at the farm could help to limit nutrient leaching and allow stover collection. With the very high nitrogen demand in a C-C system, however, it is not likely that management practices could be sufficient to overcome the detrimental effects of eutrophication resulting from the high leaching rates.

The high nitrogen use in the C-C scenario also increases the use of fossil fuels for fertilizer production and associated emissions of species contributing to acid rain and global warming (GWP) potentials. The increase in acidification potential (+6% over the base case) is attributed to increases in NO production in soils with increased FN use and increased NO_x from fossil fuel consumed to generate the increased energy necessary for fertilizer manufacture. Increases in the global warming potential (+71% over the base case, not including benefits of carbon sequestered in soil or crop) are attributed to methane emissions from natural gas used in FN manufacture and N_2O emissions from nitrification of the additional FN.

Scenario 2 (C-S) actually reduces the global warming potential relative to the base case (-3%) and has essentially no impact on acidification. The reduction in the GWP is related to the reduced level of soil mineralization with no till and the resulting reduction in N_2O emissions.

Human and ecological toxicity are also important impact categories for agricultural activities. Nitrates can contribute to human health impacts through drinking water from ground or surface water sources. Herbicides, which were not considered here, are as effective in waterways in killing plant life as they are on fields. The eco-toxicity of these agrichemicals should also be considered to provide the most comprehensive understanding of the environmental impacts associated with bio-based products. It is expected that higher herbicide application rates will be used with the no till practices required for sufficient stover collection.

Based on the results presented thus far and the acknowledged limitations of the model – the following recommendations for continued analyses can be made:

- Consider stover only from C-S rotations in further analyses
- Improve the erosion modeling to incorporate variability with rainfall. This could help to improve leaching models by quantifying changes in sediment-bound nutrient loads in a NT versus conventional till system.
- Use the framework developed here that utilizes water quality data to generate leaching models for herbicides. LCI data for herbicide manufacture will also need to be developed.
- Utilize additional resources to improve predictions of nutrient fate in a NT versus CT system and perform a sensitivity analysis on these processes.
- Compare results to related studies and guidelines currently used to define the acceptability of bio-based products, especially for federal procurement.

The generation of this life-cycle data was required to provide a more comprehensive understanding of the environmental impacts associated with increased use of biomass for fuels and other products. The LCA quantifies impacts on very different components of the environment, but does not judge which of these components are more important. The LCA results are necessary, but not sufficient to allow decisions regarding future energy sources to be made. In the United States, the goal of current policies is to reduce our dependence on imported energy. The LCA results can be used to quantify the environmental benefits and detriments associated with this goal and can help identify key concerns that should be addressed as we move forward with this goal (e.g., improved nutrient management to reduce water quality degradation). At some point, a balance needs to be defined between the various goals of different interest groups in a manner that then overall environmental impact from biomass fuels and products is considered acceptable, leading to a sustainable materials and energy source for the future.

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Appendix A – Definition and Quantification of Variables Used in Nutrient Flow Model

Table A-1: Parameters describing flows through the system[a]

Symbol	Definition	Sources of information
L_{NH3}	Total ammonia load added to the system	Calculated, equation [13]
L_f	Total ammonia fertilizer load	Calculated, eqn. [14]
L_f^P	Total phosphorus fertilizer load	Calculated, eqn. [39]
L_f^K	Total potassium fertilizer load	Calculated, eqn. [44]
$L_{atm,i}$	Atmospheric deposition of i=NH3/NH4+ or NO3/HNO3	Calculated, eqn. [15] and [22]
$L_{soil\ min}$	NH_3-N load generated by soil mineralization	Calculated, eqn. [16]
$L_{crop\ min}$	NH_3-N load generated by crop residue mineralization	Calculated, eqn. [17]
L_{vol}	Total NH_3-N volatilized from ammonia fertilizer	Calculated, eqn. [18]
L_{senes}	Total NH_3-N lost during plant senescencing	Calculated, eqn. [19]
L_{ni}	nitrate load generated though nitrification	Calculated, eqn. [20]
L_{NO3}	Total nitrate load from all sources	Calculated, eqn. [21]
L_{harv}	Nitrogen removed from the system with the crop and stover harvest	Calculated, eqn. [23]
$L^P{}_{harv}$	Phosphorus removed from the system with the crop and stover harvest	Calculated, eqn. [40]
$L^K{}_{harv}$	Potassium removed from the system with the crop and stover harvest	Calculated, eqn. [45]
L_{res}	Nitrogen bound in crop residues left on field	Calculated, eqn. [24]
L_{SW}	Nitrogen leached to surface water via runoff and tile/base flow drainage	Calculated, eqn. [25], [38]
$L^N{}_{SW,C}$	Nitrogen leached to SW that is allocated to corn, allocation method 2	Calculated, eqn. [47]
$L^P{}_{SW}$	Phosphorus leached to surface water via runoff and tile/base flow drainage	Calculated, eqn. [41], [43]
L_{MS}	Nitrogen discharged to the Mississippi River	Calculated, eqn. [37]
L_{GW}	Nitrate leached to groundwater systems (below zone influenced by tiles)	Calculated, eqn. [26]
L_{de}	Total nitrate lost via denitrification	Calculated, eqn. [27] or [28]
$L_{de,i}$	Nitrate lost due to denitrification in a specific compartment I=dr, ri, gw, Gu)	Calculated, eqn. [29a – 29e]
L_{imm}	Nitrate immobilized as organic-N in SOM	Calculated, eqn. [30]
L_{fix}	Nitrogen added to the system through fixation by soy	Calculated, eqn. [31] or [32]
L_{er}	Nitrogen transported to surface water through soil erosion	Calculated, eqn. [33]
L_{N2O}	Nitrous oxide emissions	Calculated, eqn. [34]
L_{NO}	Nitrogen oxide emissions	Calculated, eqn. [35]
L_{N2}	Nitrogen emissions	Calculated, eqn. [36]

[a] all flows expressed as N, P or K (mt yr^{-1})

Table A-2: Rates required for nutrient flow calculations (kg ha^{-1} yr^{-1} unless otherwise noted)

Symbol	Definition	Sources of information
$N_{f,i}$	Annual historical fertilizer rate per area fertilized, i=corn or soy	USDA NASS data base and USDA Census reports, IA average values (http://agcensus.mannlib.cornell.edu/, http://www.usda.gov/nass/pubs/histdata.htm (Higher values - +45 kg/ha used for C-C systems, Gentry et al., 2001)
$N_{f,st}$	Additional NH_3-N added to replace amount removed with corn stover	Calculated, eqn. [1b]
$N_{senes,i}$	Ammonia loss through plant senescencing (i=corn, soy)	Literature values used initially (Burkart and James, 1999, Goolsby et al., 1999) ($N_{senes,ci}$ =50-60; $N_{senes,soy}$ = 45). Calibrated values much lower ($N_{senes,ci}$ =30; $N_{senes,soy}$ = 20)
H_i	Harvest yield for i=corn or soy	USDA NASS data base and USDA Census reports for each county (http://agcensus.mannlib.cornell.edu/, http://www.usda.gov/nass/pubs/histdata.htm, converted with bulk densities – 56 lb/bu, corn and 60 lb/bu soy (USDA PLANTS database)
$H_{res,min}$	Stover that must remain on the field	Sheehan et al. (2002) and Nelson (2004) – estimates for C-C and C-S rotations for IA counties, average rain
Q_{rain}	Annual rainfall (mm yr^{-1})	Data for centroid of each county determined from PRISM Spatial Climate Analysis Service, 1988 – 2000. Varies by county, year http://www.ocs.orst.edu/prism/
$N_{fix,N}$	Soy nitrogen fixation rate	Jordan and Keller (1996) $N_{fix,N}$ =78 (range = 15-310)
N_i	Grain yield (i=corn, soy)	USDA NASS Quick Stats database for bu/ac yields – varies by country, year (http://www.nass.usda.gov:81/ipedb/). With bulk densities use for conversion (56 dry-lb corn grain/bu; 60 dry-lb soy/bu) (USDA plants database)
N_{st}	Stover harvest yield	Calculated based on required residue to meet tolerable erosion (Sheehan et al., 2002; Nelson, 2004), varies by county, year

Table A-3: Other parameters

$A_{i,j}$	Crop area (ha) of i = corn or soybeans, j=planted, harvested	USDA NASS data base and USDA Census reports for each county (http://agcensus.mannlib.cornell.edu/, http://www.usda.gov/nass/pubs/histdata.htm
A_{ws}	Area of watershed (ha)	Approximated by county boundaries. County areas from US Census (2000). http://quickfacts.census.gov/qfd/maps/iowa_map.html
C_i	Concentration of i=ammonia, nitrate or sulphate in wet deposition (mg/L)	Concentrations from historical data at three locations near Eastern Iowa, averaged over system for each year. National atmospheric deposition program, http://nadp.sws.uiuc.edu/
M_{soil}	Mass of soil in 1 ha, 30 cm deep	Schepers and Moiser, 1991 (assuming bulk density = 1480 kg/m^3)

Table A-4. Fractions describing distribution of nutrient flows (unitless)

Symbol	Definition	Value	Range	Source of information
$f_{f,i,j}$	Fraction of crop acreage (i=corn or soy) that fertilizer (j=N, K, P) is applied	Historical data used (USDA NASS)		
$f_{i,j}$	Fraction of crop (i=corn, soy, stover) that is nutrient (j=N, K, P)	See Table A-4		
f_{vol}	Fraction of ammonia fertilizer that volatilizes	0.03	0.0 – 0.15	Meisinger and Randall (1991) for injected anhydrous ammonia. Puckett et al. (1999) 5-15%, for AN NH3.
$f_{DD,NH3}$	Ratio dry deposition to wet deposition, NH3/NH4	0.11	0.13 – 0.7 (all N species)	Goolsby et al., 1999 (MRB)
$f_{DD,NO3}$	Ratio dry deposition to wet deposition, NO3, HNO3	0.41	0.13 – 0.7 (all N species)	Goolsby et al., 1999 (MRB)
f_{ni}	Fraction of ammonia that is nitrified	0.95	0.90 – 1.00	deVries et al. (2003) value for moist to wet Loess soils
$f_{ni,N2O}$	Fraction of nitrogen that is nitrified and emitted as N_2O	0.005	0.005 – 0.02	deVries et al. (2003)
$f_{ni,NO}$	Fraction of nitrogen that is nitrified and emitted as NO	0.01	0.01 – 0.03	deVries et al. (2003)
f_{om}	Fraction of soil that is organic matter	County-specific data from Brenner et al. (2001) used (average = 2.7%, range 2.4 – 3.1%) Goolsby et al. (1999) used 3.5% for IA average value		
$f_{om,N}$	Fraction of organic matter that is nitrogen	0.03		Meisinger and Randall, 1991; Schepers and Moiser, 1991
$f_{min,c(t-1)}$	Fraction of soil org-N that is mineralized in a year after corn was grown	0.014	0.01 – 0.03	Range from Schepers and Moiser, 1991, specific values from Gentry et al. (2001) data. Lower values used for NT.[a]
$f_{min,soy(t-1)}$	Fraction of soil org-N that is mineralized in a year after soy was grown	0.02	0.01 – 0.03	Range from Schepers and Moiser, 1991, specific values from Gentry et al., data.
HI_c	Harvest index for corn – amount of corn stover generated per harvest of corn grain	1.0	0.75 – 1.4	Gupta et al. (1979) ; range from Pordesimo et al., 2004 (higher values for early harvest). Linden et al. (2000) 0.783 ±0.11
HI_{soy}	Harvest index for soy – amount of soy straw generated per harvest of soybean harvest	1.5		Gupta (1978)

$f_{i,SW}$	Fraction of total fertilizer (i=N, P) that is leached into tiles and drains	See regression analyses, Table 10, eqn [43]		
$f^{P}_{sw,NT}$	Factor to account for increased leaching of TP from NT practices	1.6	0 – 5.4	Gaynor and Findlay (1995); McIsaac et al. (1995)
$f_{NO3,gw}$	Fraction of net anthropogenic nitrogen that is leached into groundwater	0.03	--	Howarth et al., 1996
f_{de}	Fraction of anthropogenic nitrate that is denitrified	0.1	0.06 – 0.25	Meisinger and Randall, 1991; Howarth et al., 1996
$f_{de,s}$	Fraction of available soil nitrate that is denitrified	0.5	0.4 – 0.95	deVries et al. (2003), higher values for wetter soils
$f_{de,gw}$	Fraction of nitrate leached to GW that is denitrified	0.4	0.4 – 0.95	Galloway (2003); deVries et al. (2003), higher values for wetter soils
$f_{de,dr}$	Fraction of nitrate in tile drains that is denitrified	0.0	0.4 – 0.95	deVries et al. (2003) (loss already integrated into L_{sw} calculation)
$f_{de,ri}$	Fraction of nitrate discharged to rivers that is denitrified	*	0.3 – 0.7	deVries et al. (2003) * see narrower application of ranges based on river flow rate (Table 8)
$f_{de,Gu}$	Fraction of nitrate discharged to the Gulf of Mexico that is denitrified	0.9	> 0.8	deVries et al. (2003)
$f_{de,NO}$	Fraction of nitrogen, which is denitrified, that is emitted as NO	0.01	0.01 – 0.02	deVries et al. (2003)
$f_{de,N2O}$	Fraction of nitrogen, which is denitrified, that is emitted as N_2O	0.015	0.01 – 0.06	deVries et al. (2003), higher values for peat soils
f_{fix}	Fraction of N in soy plant that is attributed to N_2 fixation	0.65	0.5-0.8	Range for soy with 60-100 kg/ha available N in soil. Lower end of range for f_{om} > 0.03 (Meisinger and Randall, 1991)
$f_{i,imm}$	Fraction of nitrate (i=fertilizer or atm dep.) that is immobilized as org-N	0.25	0.4±25-50%	Goolsby et al. (1999)
$f_{er,sed}$	Fraction of soil eroded that is delivered to surface water	0.51		Sheehan et al. (1998)

[a] lower mineralization rates for no till result in smaller fraction of total nitrogen in SOM being released by mineralization. Based on data quantifying increased carbon sequestration rates in Iowa for NT (Brenner et al., 2001) (2.42 million mt C sequestered in Iowa with NT, 1.57 million mt C sequestered with conventional till, suggests that 54% difference to NT. $f_{min,i}$ reduced by this factor for NT.

Table A-4.	Nutrient content in crops[a]		
Plant material	frac N	frac P	frac K
corn grain	0.0150[b]	0.00267[c]	0.0034
corn stover	[d]	0.00010	0.0150
soy bean	0.0650[e]	0.00653 [c]	0.0154
soy straw	0.0085[e]	0.00060	0.0057

[a] from USDA, NRCS, The PLANTS Database unless otherwise noted (Version 3.5, http://plants.usda.gov. National Plant Data Center, Baton Rouge, LA, 2004)

[b] 0.015 used by Sheehan et al. (2002), typical range 0.0135 – 0.0175 (Meisinger and Randall, 1991), 0.0164 in PLANTS database

[c] value in table from Goolsby et al. (1999). These lower values used to help with mass balance. Values from PLANTS data base are $f_{c,P}$ = 0.0031 and $f_{s,P}$ = 0.0067.

[d] 0.0045 used by Sheehan et al. (2002), 0.0098 in PLANTS database. Regression equation used here to determine variability as a function of rainfall. Regression equation determined from data and approach in Balcom et al. (2003):

$$f_{st,N} = \frac{\left(13.4 * \exp(-0.009 \cdot \mathrm{rain}) + 1.3\right)}{1000} \qquad [48]$$

values ranged from 0.0017 to 0.0061 with median =0.0025.

[e] value from Meisinger and Randall (1991). Values from PLANTS data base, $f_{s,N}$=0.0657; $f_{straw,N}$=0.0.0083

Appendix B – Files comprising LCA model

File Name	Description
summary - input.xls	Includes most input parameter data sets (crop yields, fertilizer use, rain, etc)
nitrate model-C-S.xls	Nitrogen flow model for Scenario 1 -base case (C-S, no stover)
nitrate model-C-S-stover.xls	Nitrogen flow model – Scenario 2 (C-S w/ stover)
nitrate model-C-C-stover.xls	Nitrogen flow model – Scenario 3 (C-C w/ stover)
phosphate.xls	Phosphorus flow model for Scenario 1 -base case (C-S, no stover)
phosphate-stover.xls	Phosphorus flow model – Scenario 2 (C-S w/ stover)
phosphate-C-C-stover.xls	Phosphorus flow model – Scenario 3 (C-C w/ stover)
potassium.xls	Potassium flow model for Scenario 1 -base case (C-S, no stover)
potassium-C-S-stover.xls	Potassium flow model – Scenario 2 (C-S w/ stover)
potassium-C-C-stover.xls	Potassium flow model – Scenario 3 (C-C w/ stover)
fuel use.xls	Fuel use calculations – on farm
nutrient impact.xls	Eutrophication impact analysis for all scenarios
flows from DEAM.xls	Downloaded files from DEAM database and reorganization for models used here
overall LCA.xls	Summary of all LCI flows, all scenarios, with impact analysis
summary-graphs.xls	Includes most final graphs used in report, all linked to original worksheets where values calculated

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14. ABSTRACT *(Maximum 200 Words)*
Fertilizer use can cause environmental problems, particular eutrophication of water bodies from excess nitrogen or phosphorus. Increased fertilizer runoff is a concern for harvesting corn stover for ethanol production. This modelling study found that eutrophication potential for the base case already exceeds proposed water quality standards, that switching to no-till cultivation and collecting stover increased that eutrophication potential by 21%, and that switching to continuous-corn production on top of that would triple eutrophication potential.

15. SUBJECT TERMS
eutrophication; stover; life-cycle analysis; nitrogen; phosphorus; no-till; continuous corn; fertilizer; ethanol

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