A Review of BMPs for
Managing Crop Nutrients and Conservation Tillage
to Improve Water Quality

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Introduction

Nutrients

Nutrients are essential for the growth of all living organisms. Modern agriculture depends on nutrients supplied by many sources. Sources of nutrients for crops include the mineralization of soil organic matter, animal manure, sewage sludge, commercial fertilizers, nitrogen fixed by legumes, nitrogen contained in irrigation water, and atmospheric deposition. Research conducted over many decades has aided farmers in the efficient use of added nutrients through techniques such as soil testing and nutrient placement and timing. Until recently, most research and education was aimed at helping farmers determine economically optimal nutrient application amounts and methods. Today, we are more aware of the adverse off-site impacts that nutrients may have when they leave agricultural fields with surface runoff or leaching and enter surface or ground water in excessive amounts. Nutrient losses, which may be unimportant from an economic standpoint, may still cause impacts harmful to aquatic ecosystems, or harmful to human health. This increased knowledge and concern about the adverse impact of excessive nutrients on water quality has forced a reexamination of agricultural nutrient management practices and their impacts on water. New water quality standards and regulations are being developed that may force changes in nutrient management, especially in areas where waters have been determined to be impaired by excessive nutrients. As some proposed nutrient standards for surface water are far lower than current nutrient levels, agriculture and the rest of society face a major challenge to reduce nutrient losses.

Agricultural researchers have responded to concerns about nutrient impacts on water quality by investigating the use of Best Management Practices or BMPs to help reduce nutrient losses. The two nutrients which have the greatest impact on water quality are nitrogen (N) and phosphorus (P). Due to potential human health impacts of excessive levels of the nitrate (NO_3^-) form of nitrogen in drinking water and the existence of nitrate drinking water standards, many farmers are aware of the impacts of excessive nitrogen losses. Nitrogen management educational efforts have been underway for many years and some water quality improvements as a result have been measured. However, both excess N and P can adversely affect aquatic systems by causing undesirable growth of algae and other aquatic plants. These impacts may be experienced in nearby waters or as far away as the Gulf of Mexico. It is the threat of adverse impact of excess N and P on aquatic systems that is driving new water quality standards and efforts to curb nutrient losses.

Conservation Tillage

Conservation tillage is a BMP effective in reducing the loss of many surface water contaminants, including sediment, pesticides, and nutrients. However, the effectiveness of conservation tillage in reducing nutrient losses depends on many factors, such as methods and timing of nutrient applications, and local soils, topography, hydrology, and climate. Under certain conditions, some forms of conservation tillage may be less effective in reducing loss of specific nutrient forms or even increase losses. For example, surface application of manure or fertilizer to no-till fields may increase runoff losses of soluble P, the form most available to aquatic plants. While conservation tillage is an effective BMP to reduce nutrient losses, clearly additional BMPs will be needed, and some changes in tillage practices may be warranted in some situations. Fortunately, many BMPs are available to help design crop and livestock production systems that are economically sustainable and reduce nutrient losses. This publication will review research on nutrient management BMPs for the two nutrients of major concern, N and P, with an emphasis on integrating BMPs with conservation tillage. This publication should be helpful to agricultural professionals advising farmers on nutrient management practices and those writing farm-specific nutrient management plans. As specific nutrient tests, application methods and other practices must be calibrated to local conditions, and location-specific regulations may apply, always consult with Extension and other state and local resources.

Behavior of Nutrients in Soil and Water

Nitrogen

Nitrogen can follow many chemical pathways in soil and water, making it difficult to trace. N is continually cycled among plants, soil organisms, soil organic matter, and the atmosphere, as shown in Figure 1. At any given time most of the N in the soil is contained in soil organic matter (decaying plant and animal tissue) and the soil humus. N is slowly released as soil microbes decompose or mineralize the organic matter. Mineral soils contain up to 6,000 lbs/ac (6,700 kg/ha) of organic N. (Bundy 1985).

Organic N occurs as particulate matter, in living organisms, and as detritus. It occurs in dissolved form in compounds such as amino acids, amines, purines, and urea. Mineralization converts organic N into ammonium (NH_4^+), which can be taken up by plant roots. The ammonium ion is positively charged and thus is held by negatively charged clay particles and organic matter, preventing it from leaching with percolating water.

Soil bacteria convert ammonium to nitrate (NO_3^-) through the process of nitrification. *Nitrosomonas* bacteria mediate the conversion of nitrate to nitrite (NO_2^-), which is then quickly converted to nitrate by *Nitrobacter* bacteria. Nitrate is readily taken up by plant roots and is often the major form of N utilized by crops. Because nitrate is negatively charged, it enters the soil solution and is subject to leaching. Nitrification can occur rapidly in warm, moist, well-aerated soils, changing the ammonium form of N commonly found in fertilizers to the nitrate form within one to two weeks after application. Nitrification ceases at temperatures below about 50° F. Thus, application of ammonium fertilizers to soils below 50° F allows the ammonium to remain in its positively charged, immobile form until soil temperatures increase.

Denitrification is the bacterial conversion of nitrate to elemental nitrogen (N_2) or nitrous oxide (N_2O) gasses, which are lost to the atmosphere. These forms of N are unavailable to plants. Denitrification reduces nitrogen availability to crops but does not threaten water quality. Because nitrous oxide is a greenhouse gas, denitrification has climate change implications (Bouwman et al. 1980). Denitrification occurs in poorly aerated, water-logged soils. This process is rapid, so that if water stands on the soil for 2 to 3 days during the growing season, much of the nitrate – N will be lost by denitrification (Bundy 1985).

Immobilization includes processes by which ammonium and nitrate are converted to organic N, through uptake by plants and microorganisms, and bound up in the soil. Adding carbon rich crop residues to the soil causes temporary immobilization of N when bacteria take up ammonium and nitrate as they decompose crop residues. As crop residues decompose, nitrogen is again released. The speed of release varies with climate, with higher soil temperatures speeding release. In Wisconsin, release of immobilized N begins about one month after tilling crop residues into the soil (Bundy 1985).

Volatilization losses of ammonia gas to the atmosphere can occur when manure, urea fertilizer or solutions containing urea are surface applied and not incorporated into the soil. Volatilization losses are greatest with high temperatures and lack of rain after application, and where surface crop residue is present. Injection of solutions containing ammonia and incorporation of manure and urea fertilizers greatly reduces volatilization losses. Some ammonia lost to the atmosphere returns to the soil through precipitation. Precipitation also carries N from industrial and automobile emissions and nitrate formed by oxidation of nitrogen gas by lightning. Rain, snow, and dry fall deposit an average of 10 to 20 lb/acre (11 to 22 kg/ha) of available N per year in Wisconsin (Bundy 1989). In North Carolina (Whitall and Paerl 2001) 24% of new N flux to the Neuse River Estuary was contributed by rainfall. Previous studies (Paerl 1985) showed that from 20 to >40% of new N entering U.S. east coast estuarine and coastal waters was from atmospheric deposition.

Symbiotic N fixation converts atmospheric N into plant available N forms. *Rhizobia* bacteria living in the roots of legumes like soybeans, alfalfa, and clovers make N available to their host plants. As the crops decompose, N is made available to succeeding crops. Small amounts of N are also fixed by free living organisms in the soil such as *Azotobacter* and blue-green algae.

Because of its mobility, the nitrate form of N usually accounts for most N reaching surface or ground water. As ammonia is held on soil particles, little leaching occurs, but erosion can carry ammonia to surface water. Erosion and runoff can also carry organic forms of N to lakes and streams where later conversions can release plant available N.

Leaching of nitrate to groundwater can occur when nitrate in excess of crop needs is present in the soil solution and water percolates through the soil. Risk of leaching is greatest on coarse textured soils, with shallow aquifers being most vulnerable. High rainfall or irrigation in excess of crop needs increase leaching risk. In some soils, denitrification occurs in the subsoil, reducing nitrate available to leaching.

Because nitrate is mobile, it readily moves into the soil with rainfall or irrigation, rather than running off the surface of fields. In many settings surface runoff contains little nitrate. Exceptions can occur, such as when fertilizer or manure are applied to frozen soil, or heavy rains occur soon after application. The way nitrate usually enters streams and lakes is to first leach to shallow groundwater and then move laterally with natural subsurface flow or through drainage tiles (Figure 2). Areas that have been extensively tiled often have greater nitrate losses to surface water than untiled areas. In an Iowa study, at least 95% of nitrate-N percolating through tilled soils was intercepted and discharged to surface water by drainage tiles (Hatfield et al. 1998). Fausey et al. (1995) estimated that 37% of Cornbelt and Great Lakes cropland is artificially drained by surface channels, subterranean tiles, or a combination of the two. The subsurface flow pathway of nitrate to surface water is important to understand in selecting appropriate BMPs. Reducing surface runoff will not necessarily reduce nitrate losses to surface water.

The primary N fertilizers used in the U.S. are anhydrous ammonia, urea, ammonium nitrate, and urea-ammonium nitrate (UAN) solutions. As previously discussed, the ammonium and nitrate forms of N are plant available. Urea must be first hydrolzyed, or decomposed, by the enzyme urease to the ammonium form before it can be utilized by plants. Ammonium is strongly held by soil particles, while urea and nitrate are soluble and subject to leaching. Ammonia (NH_3) is volatile and can be lost to the atmosphere. Ammonia volatilization can be significant with surface application of fertilizers containing urea, especially when applied to large amounts of crop residue.

Ammonia losses may also occur from surface manure applications, or when application slots fail to close properly behind anhydrous ammonia applicators.

Phosphorus

Phosphorus (P) undergoes many transformations in the soil which affect its availability to crops and its potential to be lost to water. P exists in both organic and inorganic forms. Organic P consists of undecomposed plant and animal residues, microbes, and organic matter in the soil. Inorganic P is usually associated with aluminum (Al), iron (Fe), and calcium (Ca) compounds of varying solubility and availability to plants. P is added to soils so that there are adequate levels for optimum crop growth. However, P can be rapidly converted in the soil to forms unavailable to plants. This "fixed" P can be slowly converted to "labile" or available forms, but this conversion is usually too slow to meet crop needs. Agronomic soil tests have been developed to determine the amount of plant available P in the soil and how much fertilizer or manure should be added to meet desired crop yield goals.

There are over 200 forms of naturally occurring P minerals in the soil (Sims 1998). The most common P minerals are: 1) apatite (calcium phosphate), which is found in unweathered and moderately weathered soils; and 2) iron and aluminum phosphates, which are found in highly weathered soils. P also occurs in organic forms. Commercial P fertilizer is apatite treated with sulfuric or phosphoric acids to increase the solubility of P.

In the soil solution, P is present as either $H_2PO_4^{-1}$ in acid soils (monovalent) or HPO_4^{-2} in alkaline soils (divalent). P enters the soil solution by one of the following processes:

- Dissolution of primary minerals

- Dissolution of secondary minerals
- Desorption of P from clays, oxides, and minerals
- Biological conversion of organic P to inorganic forms

P available for crop uptake (or available to aquatic organisms when P reaches water bodies) is called bioavailable P and consists of dissolved P and a portion of P adsorbed to soil particles which can subsequently be released into solution. Analyzing a soil for total P content is not very useful from an agronomic standpoint, as it does not tell one how much P is available for crops. Agronomic soil tests have been developed over the years which extract all or a proportionate amount of bioavailable P from soils. By correlating these soil tests with crop responses to P additions to various soils in the field, recommendations for needed amounts of P additions for optimum crop production on various soils have been developed. As will be described later, these agronomic P soil tests do not necessarily predict P losses to water.

P losses are often measured as total P and dissolved P. Because dissolved P is available to aquatic organisms, it has the most immediate impact on aquatic systems. Sediment-adsorbed P constitutes 60-90% of P transported in runoff from cultivated land (Sharpley et al. 1992). Runoff from grass, forest, and noncultivated land carries little sediment and is usually dominated by dissolved P. While most adsorbed P is not readily available to aquatic organisms, sediment deposited in aquatic systems provides a long-term reservoir of P, as P is slowly released (McDowell et al. 2001). Because of the ability of lake and stream sediments to provide a long-term source of P even after inputs have been reduced, beneficial impacts of P loss reductions are difficult to predict (Young and

DePinto 1982; Gray and Kirkland 1986). Both total P and soluble P losses are important. The relative importance of each form may depend on local conditions. Results of P loss studies sometimes refer to algal available P, which is a combination of dissolved P and P readily desorbed from soil particles.

Surface runoff and erosion are the primary mechanisms carrying P to surface water in most settings. Desorption of P from a thin mixing zone of surface soil and vegetation releases dissolved P carried in runoff water. Eroded sediment carries adsorbed P and mineral and organic P sources. Because of the rapid reactions by which P is immobilized in soil, P leaching has until recently been believed to be of minor importance. However, under some conditions dissolved P and colloidal P can leach to natural subsurface flow or drainage tiles to reach surface water in significant amounts. This mechanism is most important in soils with large accumulations of P that saturate the sorption capacity of surface soils. Usually the P concentration of percolating water is low in P due to fixation by P deficient subsoils. However, sandy, acid organic, or peaty soils with low P fixation capacity can allow more P to leach (Sharpley et al. 1999). P leaching potential is thus greatest for certain regions like the Coastal Plains and Florida, as well as on certain tile-drained soils. Leaching of P may also be greater on some clay soils where macropores facilitate transport (Xue et al. 1998).

Occurrence of dissolved and total P in tile drainage has varied between studies. Baker et al. (1975) found little P in tile drainage in Iowa. Over 3 years, soluble P varied from 0 to 0.038 ppm and total P from 0.007 to 0.182 ppm. Only 6 out of 477 samples had total P concentrations > 0.1 ppm. In contrast, in Illinois Xue et al. (1998) estimated that

46 to 59% of the dissolved P export to the upper Embarras River was from tile drainage. Sims et al. (1998) have reviewed the importance of P loss in agricultural drainage.

On the average, only 30% of the fertilizer and feed P input into farming systems is output in crop and animal produce. Therefore, when averaged over the total utilizable agricultural land area in the U.S., an annual surplus of 30 lb/ac (34 kg/ha) exists (National Research Council 1993). This surplus P increases soil fertility and may improve future crop yields if soils are low in P. But high soil P concentrations also increase the risk of loss to water. P levels have increased in U.S. soils due to fertilizer and manure applications. David and Gentry (1999) compared P inputs in Illinois (fertilizer, feed, etc.) to outputs (grain, livestock, etc.) from 1945-1998. Large net inputs were found from 1965 to 1990, with no average net inputs since 1990. For rivers in the state, they estimated that 47% of the P load was from sewage effluent. Illinois contributed 10% of the annual P load to the Mississippi.

High livestock concentrations in some areas have led to manure applications in excess of crop nutrient needs. If manure applications are based on N content and crop N needs, excess P is applied. In the past, high P soil tests levels were not considered to cause environmental problems, so manure was commonly applied based on N levels. Application of manure to corn to meet a 200 lb/ac (224 kg/ha) N need adds 100 lb/ac (112 kg/ha) P or more (Sharpley et al. 1999). Corn removes only about 30 lb/ac (34 kg/ha) P, so about 3 times the P needed by the crop is applied with this manure rate. Such an application to a soil testing low in P could be viewed as appropriate, as it would build soil P towards optimum levels. But when excess P applications are continued for many years, soil test levels rise to excessive levels and risk of water contamination is increased.

Manure utilization has become a dilemma for farmers in some areas, as more than twice as many acres may be needed to apply manure based on P levels as N levels. Transporting manure long distances to land testing lower in P is costly.

Once P soil test levels reach excessive levels, it may take many years for P concentrations to decrease. McCollum (1991) estimated that without further P additions, 16 to 18 years of corn and soybean production would be needed to deplete a soil test P (Mehlich – 3) in a Portsmouth fine sandy loam from 100 ppm to the agronomic threshold of 20 ppm. So it is important to manage current manure and fertilizer applications to minimize both current and future P loss risks.

There has been a general increase in soil test P levels in the U.S. since World War II, as a result of P applications as fertilizer and manure. A 1989 summary of soil test values showed that in several states more than 50%, and in some states 75%, of soil test P samples tested high (Potash Phosphate Institute 1994). A 1997 soil test summary indicates that many agricultural soils remain in high and above categories (PPI/PPIC/FAR 1998). The percent of soils testing high are similar to the 1989 percentages for many states, but there are signs of a decreasing high soil test P trend in some Midwest states such as Indiana, Illinois, Iowa, Minnesota, and Ohio. Soil test P levels continued on an increasing trend in Arkansas, Wisconsin, North Carolina, and Delaware.

The presence of soils testing high in P may simply mean a crop farmer can forgo P fertilizer applications for some period of time. But such soils present a problem for livestock farmers in need of land to apply manure. Regulations or required manure management plans could limit manure applications to such land in the future.

Impact of Conservation Tillage on Nutrient Losses

Overview

Conservation tillage systems impact both soil erosion and water infiltration, which in turn can affect the runoff or leaching of N and P. The type of tillage system used also influences where nutrients are found within the soil profile and their vulnerability to loss. Systems utilizing some form of full width tillage allow the incorporation of applied fertilizers and manures, removing some nutrients from the soil surface and placing them away from overland flow which could carry them to surface water. Fertilizers and liquid manures can be injected or otherwise placed below the soil surface in any tillage system, including no-till, protecting them from runoff, but incorporation of dry manures requires some form of tillage. Continuous application of fertilizer or manure to the soil surface in no-till systems results in a stratification of non-mobile nutrients like P (Erbach 1982). Higher P concentrations at the soil surface increase the availability of P for runoff. The organic matter content of surface soils also increases with no-till (Reicosky et al. 1995). This increased organic matter could reduce runoff losses of some pollutants, like pesticides and P, by providing more adsorption capacity and causing increased infiltration, due to improved soil structure. Losses of total N could be increased relative to tilled soils, if significant runoff or erosion occurs.

The ability of conservation tillage to reduce erosion is well documented. Crop residue left on the soil surface protects the soil from the erosive impacts of rainfall and wind. Residue also slows runoff and prevents sealing of the soil surface, increasing infiltration of water. Reductions in erosion are usually proportional to the percent of the soil surface covered by crop residue. Conservation tillage often increases water infiltration. Surface roughness and surface residue are responsible for infiltration increases in conservation tillage systems utilizing some form of tillage. In no-till systems, improved soil structure and the presence of macropores consisting of worm holes, cracks and root channels, allows water to infiltrate rather than runoff when rainfall exceeds the capillary flow capacity of the soil (Fawcett and Caruana 2002).

No-till has sometimes dramatically increased water infiltration and reduced runoff. Edwards et al. (1988) compared season-long water runoff from a 0.6-acre watershed with a 9% slope that had been farmed for 20 years in continuous no-till corn to a similar conventionally tilled watershed. Over four years, runoff was 99% less under the long-term no-till. No-till has reduced runoff well even under extreme conditions. A no-till watershed on a 21% slope had almost no soil erosion and held water runoff to levels similar to a conventional tillage watershed of only 6% slope during a once-in-100 yr storm of 5 in. (12.7 cm) in 7 hr (Harold and Edwards 1972). No-till may not increase infiltration where water percolation through the soil profile is prevented by conditions such as an impervious claypan, high water table, or compaction.

Soils are classified into one of four hydrologic soil groups based on rates of water infiltration and transmission when wet. Group A soils are mainly sandy and have a high infiltration rate when wet and high rate of water transmission. Group B soils have a moderate infiltration rate and moderate rate of transmission. Group C includes soils with a slow rate of water infiltration when wet. They have a layer that impedes the downward movement of water or they are moderately fine textured to fine textured. Group D includes soils with a slow infiltration rate when wet and with very slow water transmission. This group includes clay soils, soils with a permanent high water table, soils with a claypan near the surface, and soils that are shallow over nearly impervious material. A review of published field studies which compared water runoff with no-till versus conventional tillage was conducted to determine if the ability of no-till to reduce runoff could be correlated to soil hydrologic group (Fawcett and Caruna 2002). No studies conducted on Group A soils were found. All studies (19 comparisons) conducted on Group B soils documented decreases in water runoff with no-till, with runoff averaging 56% of that with conventional tillage. For Group C soil (26 comparisons), 85% of studies documented reductions in water runoff with no-till, with runoff averaging 67% of that with conventional tillage. Only 9% of Group D soils (11 comparisons) responded to no-till with reduced runoff, with runoff averaging the same between tillage systems. Thus, no-till can not be expected to reduce water runoff on Group D soils.

Nitrogen

As nitrate is soluble and quickly moves into the soil with rainfall or irrigation, little nitrate is usually present in surface runoff. Ammonia held on soil particles and organic nitrogen can move off fields with erosion and runoff. Conservation tillage reduces runoff of these forms of nitrogen. A 97% reduction in soil loss for no-till relative to the moldboard plow resulted in a 75 to 90% reduction in total N loss for soybeans following corn and 50 to 73% reduction in total N loss for corn following soybeans (Baker and Laflen 1983). Other studies have documented reductions in N losses with conservation tillage (Seta et al. 1993; Barias et al. 1978; Johnson et al. 1979).

Because in most settings nitrate reaches streams by first infiltrating and then moving with subsurface flow, increases in infiltration caused by conservation tillage could impact both nitrate leaching and eventual movement to surface water. Many researchers have investigated the impact of no-till and other conservation tillage systems on nitrate leaching. Most studies have found little impact, with some studies finding a reduction in nitrate leaching with no-till.

Early studies in Kentucky on silt loam soils indicated that no-till might increase nitrate leaching. Thomas et al. (1973) first compared planting corn no-till into killed sod with conventionally planted corn. Soil cores were taken to 35.4 in. (90 cm) before and after June rainfall and analyzed for nitrate. More nitrate was lost from no-till plots, apparently due either to denitrification or leaching. Later, pan lysimeters were installed 39.4 in. (100 cm) deep under similar plots (Tyler and Thomas 1977). Over a 2-month period more nitrate was recovered in leachate from no-till plots than from conventional tillage plots. However, in a subsequent study using ^{15}N-depleted ammonium nitrate, tillage system did not affect the amount of N missing and presumed lost by denitrification or leaching (Kitur et al. 1984). After adding ^{15}N-depleted fertilizer to the same plots for 3 years, 71 to 75% of fertilizer was accounted for in grain, stover, and soil.

Kanwar et al. (1985) applied 134 lb/ac (150 kg/ha) of N (as KNO_3 solution) to the surface of no-till and conventionally tilled field plots in Iowa and applied two simulated rains of 5 and 2.5 in (12.7 and 6.4 cm) one day apart. Soil was then sampled down to 61 in (150 cm) and analyzed for N content. Much more N was lost from the conventionally tilled plots, presumably leached below 61 in (150 cm). In no-till plots 25.9 lb/ac (29 kg/ha) N leached from the 61in (150 cm) profile, while 109 lb/ac (122 kg/ha) leached in plowed plots. The authors attributed this large reduction in leaching with no-till to the bypassing of water through macropores in no-till plots.

Monitoring of drainage tile effluent has proven useful in investigating tillage impacts on nitrate leaching. Kanwar et al. (1988) monitored tile effluent from continuous corn plots managed with no-till and conventional tillage in Iowa. Nitrate concentrations were similar between tillage systems in the first two years. By the third year of the study, nitrate concentrations were significantly lower in no-till plots. Subsequently, these plots were monitored over an 8-year period (Kanwar and Baker 1993). Nitrate concentrations in tile effluent were consistently lower with no-till than with conventional tillage. Piezometers installed at 5 depths from 3.9 to 11.8 ft (1.2 m to 3.6 m) also showed that nitrate concentrations were lower at all depths with no-till.

Randall and Iragavarapu (1995) conducted an 11-year study in southern Minnesota investigating the impact of no-till and conventional tillage in continuous corn production on nitrate leaching by monitoring drainage tiles in a clay loam soil. Mean annual tile flow was 12.4 in (31.5 cm) for no-till and 11.0 in (28 cm) for conventional tillage. Flow weighted average nitrate concentrations were 12.0 ppm and 13.4 ppm for no-till and conventional tillage, respectively. Total losses of nitrate were 5% less with no-till. Thus, higher tile flows with no-till did not lead to greater losses of nitrate.

Kanwar et al. (1997) continuously monitored drainage tiles in a long-term tillage experiment in northeast Iowa. Moldboard plow, chisel plow, ridge-till, and no-till systems had been in place 14 years before monitoring began. Over 3 years of monitoring, nitrate concentrations in tile effluent were lower with no-till and ridge-till than with moldboard plow or chisel plow. Because tile flow was sometimes greater with no-till and ridge-till, differences in total seasonal losses of nitrate were not statistically significant for any system.

In Ontario, nitrate movement under conventional tillage and no-till were compared for 4 years on a silt loam soil (Patni et al. 1998). Nitrate concentrations in ground water were higher under conventional tillage than no-till every year and every season at depths of 3.9, 5.9, and 15.1 ft (1.2, 1.8, and 4.6m).

Nitrate concentrations in groundwater and in drainage tile effluent have been consistently lower under no-till management than with conventional tillage. However, increased infiltration with no-till may at least partially offset lower concentrations, resulting in similar masses of nitrate leaching below fields to where nitrate may reach ground water or be carried by subsurface flow to surface water. Conservation tillage is thus unlikely to have either a significant positive or negative effect on nitrate losses to water. Other BMPs will be needed to manage nitrate losses.

Phosphorus

Because total P losses in runoff are made up primarily of insoluble P carried by eroded sediment particles, conservation tillage usually reduces total P losses. Particulate P often represents 60 to 90% of the total P load of row crop runoff (Logan 1987; Sharpley et al. 1992). Conservation tillage has been an important BMP recommended to farmers to reduce P losses in specific watershed projects. For example, following wide-scale promotion of conservation tillage to reduce P loading to the Great Lakes, Baker (1993) concluded that the downward trends in total and soluble P loads from Lake Erie tributaries for the period from the late 1970s to 1993 indicated that agricultural BMPs, including conservation tillage, were effective in reducing total and soluble P export.

Controlled studies have documented the ability of various conservation tillage systems to reduce P losses. When total P runoff losses were compared between no-till and conventional tillage in Iowa (Baker and Laflen 1983), the 97% reduction in erosion with no-till resulted in an 80 to 91% reduction in total P loss for soybeans following corn. For corn following soybeans, the 86% reduction in erosion led to a 66 to 77% reduction in P loss.

Barisas et al. (1978) compared runoff losses of P in six tillage systems on 3 Iowa soils using rainfall simulation techniques. They found that as surface crop residue increased, soluble P losses increased, but because erosion was reduced by crop residue, total P losses decreased as residue increased. In a natural rainfall field study Johnson et al. (1979) compared nutrient runoff losses in ridge-till, no-till and conventional tillage. Fertilizer was surface applied before tillage. Total P losses were reduced by more than 50% by both no-till and ridge-till compared to conventional tillage. Soluble P losses were increased by no-till in one year and were similar to conventional tillage in another year. Ridge-till soluble P losses were similar to conventional tillage.

Seta et al. (1993) compared nutrient losses with the moldboard plow, chisel plow, and no-till in Kentucky (fertilizer applied before tillage). Total losses of nitrate, ammonia, and phosphate were in the order: moldboard plow > chisel plow > no-till. However, nutrient concentrations were higher with no-till. Several other studies have documented the ability of conservation tillage to reduce runoff losses of total P, while sometimes increasing soluble P losses (Romkens et al. 1973; McDowell and McGregor 1980).

Surface placement of fertilizer in no-till systems versus incorporated placement in full width tillage systems may explain much of the tendency for no-till systems to produce higher soluble P concentrations in runoff in controlled studies. Application of P in a subsurface band in no-till has prevented any increase in soluble P loss compared to conventional tillage. Andraski et al. (1985) compared runoff losses of P from four tillage systems when fertilizer was subsurface banded in all systems. No-till, chisel plow, and ridge-till systems reduced total P losses by 81, 70, and 59%, respectively, compared to the moldboard plow. Soluble P losses were also reduced by no-till and the chisel plow.

Kimmel et al. (2001) measured P runoff losses as affected by tillage system and fertilizer placement in Kansas. A chisel plow-field cultivate-disk system was compared to no-till and ridge-till, with P fertilizer either broadcast surface applied or knifed in prior to planting sorghum. Losses of total P, soluble P, and bioavailable P were measured. When the data are averaged over two years of study, knifing in fertilizer reduced losses of total P, bioavailable P and soluble P for all 3 tillage systems (Table 1.). Reductions in P losses with knifing were most evident for soluble P. Knifing reduced soluble P losses by about 75% in no-till, and ridge-till.

Table 1. Tillage and P placement effects on soluble, bioavailable, and total P loss in runoff water from sorghum grown on a silt loam soil with 1.0 to 1.5% slope in Kansas. Adapted from Kimmell, R.J., G.M. Pierzynski, K.A. Janssen, and P.L. Barnes. 2001. J. Environ. Qual. 30:1324-1330.

Tillage System	**Fertilizer Placement**	**Annual P Runoff Loss Average of 2 Years Data**		
		Soluble P	**Bioavailable P**	**Total P**
			(g/ha)	
Chisel-disk	Surface	16.0	49.5	605.0
Chisel-disk	Knifed-in	12.3	33.0	354.0
No-Till	Surface	329.0	398.5	832.5
No-Till	Knifed-in	73.5	123.5	479.5
Ridge-Till	Surface	320.5	426.0	1122.5
Ridge-Till	Knifed-in	77.5	121.5	675.5

Considering published studies, conservation tillage can be expected to consistently reduce runoff losses of total P. Losses of soluble P may be higher with no-till if P fertilizers are surface applied to no-till compared to incorporated in other tillage systems. However, subsurface banding of P fertilizer in no-till systems reduces losses of soluble P below loss levels for conventionally tilled soils with the same fertilizer application method.

BMPs for Nitrogen Management

Soil Conservation Practices

As discussed previously, conservation tillage reduces runoff losses of ammonia and organic nitrogen by reducing erosion but increases preferential flow (Power et al., 2000). Other erosion control practices such as contour planting and terracing also reduce the runoff of these forms of nitrogen. In Nebraska (Schepers et al. 1985), tile outlet terraces and sediment basins were studied. Sediment-born N and P accounted for 85% and 98% of total N and P losses in runoff from land. Because a pool which formed around riser inlets allowed sediment to settle out, these erosion control structures were effective in reducing nutrient concentrations in runoff. As these terraces increase water infiltration, they may affect the leaching of nitrate and subsequent subsurface movement to surface water. However, no studies have been conducted to investigate this possibility.

Kanwar and Colvin (1995) found that nitrate concentration in tile drains from no-till soils were less than from chisel-plowed soils, but these results have not been so consistent in other studies. After a long-term study Randall and Iragavarapu (1995) found that nitrate losses were slightly higher with conventional tillage than with no-tillage.

Several authors concluded that tillage systems have a minor effect on nitrate losses compared to other N management practices (Randall and Mulla, 2001), crop rotation (Weed and Kanwar, 1996), and weather (Kitchen et al., 1998).

Nitrogen Rates

The rate of N applied, whether the source is fertilizer, manure, or any other source is one of the most important factors affecting potential nitrate losses to water (Power and

Schepers, 1989; Meisinger and Delgado, 2002). Nitrate that is not taken up by crops is subject to loss and may reach groundwater or surface water. Most of the subsurface drainage loss of nitrates occurs between November and May (Cambardella et al., 1999; Meisinger and Delgado, 2002).

When nitrogen was applied at low (60 or 51 lb/ac) medium (102 or 120 lb/ac) and high (154 or 180 lb/ac) rates to corn grown in a corn-soybean rotation and nitrate leaching losses measured over 4 years, N losses averaged 26, 31, and 43 lb N/ac/yr (29, 35, and 48 kg N/ha/yr), respectively, for the low, medium, and high N rates (Jaynes et al. 2001). Andraski et al (2000) reported that average soil water nitrate concentration at a 120 cm depth were more than 20 mg L^{-1} where the N rate applied exceeded the economically optimal N rate by 50 kg N ha^{-1} or more.

Extension specialists at state universities have developed N recommendations and formulas based on crop responses measured in historical studies and N removal rates of crops. Formulas used to determine N fertilizer needs start with an expected yield goal and the amount of N needed to produce the yield. Use a reasonable method to determine expected yields to avoid overfertilization. The ability of soils to produce N through mineralization (most related to organic matter levels) is often taken into account only to a limited degree. N credits from sources such as legume crops grown in rotation, animal manure or other organic wastes, and N in irrigation water are then subtracted. Soil tests may also be used to measure available N.

Nitrogen Credits

Legumes. Legumes can provide significant amounts of N to crops grown in rotation, with a good stand of alfalfa sometimes providing all the N needs of a corn crop

(Bundy and Andraski, 1993). Table 2 shows examples of state Extension nitrogen credits for previous legume crops.

Table 2. Nitrogen credits for legume crops.

Crop	**Nitrogen Credit** **lb N/ac**
Forages	
Alfalfa	
> 50%	80-120
25-50%	50-80
< 25	0-40
Red Clover and Trefoil	
> 50%	60-90
25-50%	40-60
< 25%	0-30
Soybeans	1 lb N/ac for each bu/ac harvested up to 40 lb N/ac
Green Manure Crops (plowed down after growing season of the seeding year)	
Sweet Clover	80-120
Alfalfa	60-100
Red Clover	50-80
Vegetable Crops (residue not removed)	
Peas, snap beans, lima beans	10-20

Sources: Pennsylvania State University 1997. The Penn State Agronomy Guide, 1997-1998, University Park, Pa. University of Wisconsin-Extension and Wisconsin Dept. of Agriculture, Trade, and Consumer Protection, 1989. Nutrient and Pesticide Best Management Practices for Wisconsin Farms. WDATCP Technical Bulletin ARM-1, Madison, WI.

Manure Credits

N and P credits for manure, sludge and other organic wastes will be discussed in detail in a later livestock and manure management section.

Irrigation Water Credits

Irrigation water can contain significant amounts of N, especially in regions where soils are coarse and groundwater shallow. In the Central Platte River Valley in Nebraska, ground water used to irrigate corn contributed an average of 41 lb N/ac (46 kg N/ha), nearly one-third of the N fertilizer requirement (Schepers et al. 1986). Ferguson et al. (1991) evaluated irrigation and N practices in 79 commercial farms and found that N carried with the irrigation water supplied 19 lb N/ac (21 kg N/ha). In Wisconsin, ground water used to irrigate potatoes contributed an average of 51 lb N/ac (57 kg N/ha), or 25% of the N added as fertilizer (Saffigna and Keeney 1977). Taking credit for N in irrigation water saves on fertilizer costs and prevents overfertilization. The amount of N available from irrigation water can be calculated by multiplying the nitrate-N concentration (in ppm) times 0.23 for each acre-inch of water applied. Credit for N in irrigation water should be estimated based on the amount of irrigation water that will be applied in a year with above-average rainfall, so that N will not be deficient if less irrigation water is used than anticipated. Table 3 shows nitrogen contributions from irrigation water for 4 regions of Nebraska, based on projected net irrigation amounts.

Table 3. Nitrogen contribution from irrigation water for regions of Nebraska.

Nebraska Region	Net Irrigation	Irrigation Water Nitrate-N (ppm)						
		10	15	20	25	30	35	40
	(inches)	lb N/acre						
East	6	14	20	27	34	41	48	54
Central	9	20	30	41	51	61	71	82
West	12	27	41	54	68	81	95	108
Panhandle	15	34	51	68	85	102	118	135

From Fertilizer Nitrogen Best Management Practices, University of Nebraska – Lincoln Publication G94-1178-A.

Soil Testing Procedures

While soils have routinely been tested for non-mobile nutrients such as P and K in order to determine fertilizer application rates, soil testing for N has been problematic in some regions. Much of the available N is often present in the nitrate form, which is mobile in the soil and subject to other losses such as dentirification. Measured available N in a fall or spring soil test may not be available by the time crops need it if heavy rains cause leaching or denitrification losses. Preplanting soil tests, especially when taken at deeper soil depths, have been reliable in the relatively dry climate of the western corn belt. N present in the soil due to carryover from previous applications and mineralization of soil organic matter can be measured and subtracted from N recommendations to determine fertilizer needs. However, in more humid eastern regions, preplant N soil tests have generally not been used due to the likelihood of significant N losses occurring between soil testing and crop emergence. As a consequence, in some years when significant carryover of N occurs, N fertilizer rates have not been adjusted downward, and excessive N rates have been applied, risking increased nitrate loss to water.

Recent research has developed modified soil testing procedures to improve the reliability of soil tests in predicting N available to crops during the growing season. Usually soil samples are collected within about a month after crop planting, so that post planting N applications can be made if needed. Often these tests are referred to as “late spring” or “pre-sidedress nitrogen” tests (PSNT). Magdoff et al. (1984) in Vermont developed an N soil test for corn which involved sampling soil to a 12 in (30 cm) depth when corn was about 12 in (30 cm) tall. This procedure has since been tested and adapted to other areas. By delaying soil tests to this time, a reasonable estimate of the amount of

N released by mineralization can be made and any losses in available N through leaching, denitrification, or immobilization which have occurred prior to testing will be accounted for.

Studies in several states have shown that when nitrate-N concentrations in the surface 12 in (30 cm) of soil are greater than 20 to 30 ppm, when corn is 8 to 12 in (20 to 30 cm) tall, the probability of corn yield response to additional N application is low (Blackmer et al. 1989; Fox et al. 1989; Magdoff et al. 1990; Meisinger et al. 1992; Klausner et al. 1993). It has been more difficult to use the test to determine rates of fertilizer needed by crops, and research is ongoing to correlate PSNT results to optimal fertilization rates when deficient N concentrations occur.

Several studies have documented the effectiveness of the PSNT in reducing nitrate leaching losses. In Connecticut, Guillard et al. (1999), compared three N fertilizer regimes in corn production: PRE, 175 lb N/ac (196 kgN/ha) applied preplant; PSNT-1, 80 lb N/ac (90 kg N/ha) applied preplant and any remaining N needs estimated by PSNT [0 lb/ac in one year and 40 lb/ac (45 kg/ha) in the second year]; PSNT-2, no preplant N and all N needs estimated by PSNT (30 lb/ac or 34 kg/ha in one year and 110 lb/ac or 123 kg/ha in the second year). Corn yields did not differ between treatments. As measured by lysimeters, flow weighted nitrate-N concentrations were from 17.4 to 22.3 ppm for the PRE treatment. Nitrate concentrations were < 8.0 ppm for both PSNT treatments. Losses of nitrate-N as a percent of N applied were 20%, 10%, and 12% for PRE, PSNT-1, and PSNT-2, respectively.

In Iowa, Kanwar et al. (1996) reported that the ue of PSNT reduced nitrate loss to subsurface drainage tiles compared to a preplant application of 112 kg N ha^{-1}.

Sogbedji et al. (2000) measured nitrate leaching losses from tile-drained corn fields over 3 years on two soil types and three N fertilizer treatments: 20 lb/ac (22 kg/ha); rate based on PSNT [98 lb/ac (100 kg/ha) in most tests]; and 120 lb/ac (134 kg/ha). The first year after plowing down sod, nitrate leaching was similar with all treatments. For the subsequent 2 years nitrate leaching losses were similar for the 20 lb/ac N and PSNT-based treatments, and significantly higher for the 120 lb/ac treatment.

A 3-year study conducted in the Walnut Creek Watershed in Iowa by the National Soil Tilth Lab (Dennes et al. 2000) measured nitrate losses through drainage tile on fields where resident farmers made nitrogen management decisions (fall application from 125 to 150 lb/ac or 140 to 169 kg/ha N) compared to fields where N rates were determined by a PSNT (spring preplant N plus side-dress when needed). In the first year of study, nitrate losses were not greatly different between treatments, but by the third year, nitrate losses were reduced by 30% with the PSNT treatment compared to the standard practice of fall N application.

The PSNT was also evaluated in a 5-year study in across the Midwest that included more than 200 site-years (Bundy et al., 1999). The study showed that the PSNT successfully predicted N response in 83% of the cases with a sampling depth of 30 cm (1 ft), but it increased to 90% with deeper sampling (60 cm – 2 ft).

There are limitations to the use of the PSNT. If the test indicates that additional N is needed, fertilizer must be side-dress applied. Weather may prevent timely application and risk yield reductions. Farmers may not have available equipment and labor to apply side-dress fertilizer to all their fields in a short period of time. Soil tests must be taken during a relatively short period of time and analyses completed and reported quickly for management decisions to be made. States are still calibrating these tests to better predict amounts of N needed when the tests show deficient N levels. However, the PSNT is a tool which has often identified fields where N is already sufficient, so that planned additional N applications could be avoided, saving money and reducing nitrate leaching risk.

Because some farmers may be unwilling to use the PSNT technology due to fears of reduced yields, should the test be inaccurate, yield insurance has been proposed to encourage farmers to use the test. In Iowa (Tevis 2000) a pilot insurance program paid farmers the difference between corn yields obtained using the PSNT test to determine N rates and yields obtained with traditional N fertilization, if PSNT yields were lower. This paragraph seems out of place. In central Nebraska, the Central Platte Natural Resources District requires soil nitrate sampling on corn production fields which has helped producers identify fields with high residual soil N and adjust N management (Schepers et al., 1997).

The benefit of basing N fertilizer applications on appropriate soil tests has been demonstrated over wide areas. Schepers et al. (1993) concluded that basing N fertilizer rates on the deep soil nitrate testing recommended in Nebraska reduced ground water nitrate concentrations by about 0.5 ppm per year over a several county area in the Platte River Valley. Andraski and Bundy (2002) concluded that adjusting N rates based on PSNT or credits reduced N rates by 90 to 102 kg N ha^{-1} and increased profitability than not adjusting the N rate.

New soil testing procedures may improve N management in the future. The University of Illinois developed a soil test which measures amino sugars to predict N available from organic sources (Khan et al., 2001; Mulvaney et al. 2001). These authors showed that this soil test (ISNT) was able to separate responsive from non-responsive soils. Later work by other groups have shown additional promising results (Klapwyk and Ketterings, 2006). However, more work is needed to calibrate this soil test to different environments and then test its value as a nitrogen management tool.

Crop Testing Procedures

Analysis of crop leaf tissue for total N or nitrate has been used as a diagnostic test for a number of crops (Binford et al., 1992; Fageria and Baligar, 2005). These tests are useful only when the test has been calibrated through research trials (Blackmer 1997). This type of test is useful in detecting N deficiencies, and also in measuring how effective N management practices have been, so that corrective changes can be made in future seasons (Balkcom et al., 2003). For corn, testing ear-leaves can identify N deficiencies, but because luxury uptake does not occur, the procedure does not identify when N is excessive (Blackmer 1997).

Plant greenness is closely related to chlorophyll content, plant total N content, and potential crop yield (Schepers et al., 1992; Wood et al., 1992). Handheld chlorophyll meter units have been used to objectively assess plant greenness. These units have been valuable to detect nitrogen responsive sites, but have not been accurate enough to determine the nitrogen rate to apply (Piekielek and Fox, 1992; Varvel et al., 1997). Schepers et al. (1995) were able to use a chlorophyll meter to predict crop N status, if greenness of plants was referenced against greenness measurements in a well-fertilized plant. Water stress affects the readings of this instrument because it affects the reflectance in the red and near infra-red bands (Dinnes et al., 2002).

Because factors other than N status (such as ponding, diseases and K or Mg deficiency) can affect greenness, there have been complications in using greenness as a predictor of crop N needs. However, recent work has shown promising results in this regard (Scharf et al., 2006).

The positive results obtained with the chlorophyll meter led to other means of assessing crop greenness that could be used over large areas. Blackmer et al., (1994) found a strong relationship between corn canopy reflectance and corn greenness, and they later showed that black-and-white photographs taken with a filter could detect areas that required nitrogen fertilization (Blackmer et al, 1996). Other studies have found that aerial photographs were useful for corn nitrogen management (Tomer et al., 1997; Scharf and Lory, 2002).

Recently, other researchers have developed active hand-held sensors to measure NDVI (normalized difference vegetation index) and are actively working in developing algorithms for wheat and corn nitrogen fertilization (Raun et al., 2005). These sensors could ultimately be used with a GPS system to apply fertilizer at variable rates where N is needed.

Late Season Tests

In the past corn grain N concentration had been used as a tool to determine N sufficiency of the crop (Pierre et al.,1977), but more recently, Cerrato and Blackmer (1990) found that grain N concentration had low predictability of optimal N rates and that it was not a reliable indicator of corn N status.

For corn, analysis of the nitrate content of the lower stalk soon after black layer stage has proved useful to determine whether N was low, optimal, or excessive (Binford et al., 1992; Blackmer and Mallarino 1994). Eight-inch (20 cm) segments of stalks between 6 and 14 in (15 and 36 cm) above the soil are cut and leaf sheaths removed. Nitrate concentrations below 700 ppm indicate that N was low and additional N would have benefited yields. Nitrate from 700 to 2,000 ppm indicates that optimal N was present for the crop. A nitrate concentration of 2,000 ppm or more indicates that the crop was overfertilized and excessive N was present. By using such end-of-season checks, farmers can evaluate their N management and make changes in future years.

Chlorophyll meter readings at early dent stage of corn have been used to evaluate nitrogen sufficiency. Piekielek et al. (1995) found that this method was 93% accurate to separate N deficient from N sufficient crops. These authors reported that a relative SPAD reading of 0.93 as the critical threshold value to separate N-deficient from N-sufficient.

Fox et al., (2001) compared chlorophyll meter readings and stalk nitrate concentration of corn as tools to determine N sufficiency. Thy found that these tests had similar accuracy (>90%) and are suitable to assess corn N sufficiency.

Similarly, post-harvest soil nitrate tests are used in some regions to determine if N fertilization programs were appropriate (Sullivan 1994; Andraski et al., 2000; Cogger et al. 2001).

Timing of N Applications

Because nitrate is mobile and subject to leaching losses, and all forms of N are subject to conversion to nitrate, the longer the time that elapses between application of N and crop uptake, the greater the risk of nitrate loss. Applying N close to when the maximum N demand occurs reduces N loss risk (Meisinger and Delgado, 2002; Dinnes et al., 2002).

Fall N applications prior to spring planted crops like corn are popular in some regions. Application in the fall spreads workloads and avoids potential application delays in the spring due to weather as well as fertilizer prices are usually lower than in the spring. However, potential N losses have both economic and environmental implications. Fall and spring rainfall and soil moisture, temperature, and soil texture influence the potential for N loss. Smith and Cassel (1991) estimated N leaching depths for different fertilizer application dates. They showed that November application would leach to 1.5 m (60 in) by May whereas delaying the N application to April 1 would result in a leaching depth of 30 cm (12 in) Excessive rainfall in the fall or early spring may cause leaching of nitrate in coarse soils, or denitrification may occur in heavy, poorly drained soils. Sanchez and Blackmer (1988) found that between 50% and 64% of N applied in the fall was lost from the soil. Randall et al. (2002) showed a 36% reduction in nitrate losses from tile drainage when the fertilizer N was spring applied compared to the fall application. Table 4 shows results from a Minnesota study comparing fall versus spring N application. Higher corn yields and lower nitrate losses occurred with spring applications.

Due to risk of nitrate loss, fall application of nitrate-containing fertilizers is discouraged by many fertility specialists. Fall N applications may even be prohibited in certain areas. Fall application of anhydrous ammonia is considered an acceptable practice in some areas on non-coarse-textured soils. However, applications should be made when soil temperatures are below 50° F and trending downward. If soil temperatures are greater than 50° F, nitrification occurs, converting ammonia into the mobile nitrate form of N. Nitrification inhibitors (discussed in a later section) are sometimes recommended with fall application to reduce conversion of ammonia to nitrate.

Sidedress application, usually made about four to six weeks after planting crops, provides N just prior to the time of most rapid N uptake by crops, reducing the risk of N loss through leaching or denitrification. There are some risks with sidedress application. If sidedress applications are delayed due to weather or labor and equipment shortage, yields may be reduced. Or if N is applied to dry soil that stays dry, N may not be adequately available, reducing yields (Voss et al. 1988) Applying N in split applications involving preplant application of part of crop N needs, followed by sidedress applications, allows efficient use of applied N and reduces some risk of yield reduction should sidedress applications be delayed. Split applications and sidedress applications also allow the use of PSNT soil tests and tissue tests to better determine crop N needs.

Table 4. Effect of rate and time of N application on corn yield and nitrate N lost from tile lines in Minnesota.

Nitrogen Treatment			Annual nitrate-N
Rate	**Time**	**Yield**	**Lost in tile drainage**
(lb/acre)		**(bu/acre)**	**(lb/acre)**
0	-	66	7
120	Fall	131	27
120	Spring	150	19
180	Fall	160	34
180	Spring	168	26

Source: Buzicky, G.C. et al. 1983. Agronomy Abstr. p. 213.

Fertilizer Application Method and Placement

Surface application of urea or N solutions, especially when surface crop residues are present, risks volatilization loss (Keller and Mengel, 1986; Fox and Piekielek, 1993). Volatilization is greatest at higher temperatures. Injection of liquid fertilizers below the soil surface prevents such losses. In systems utilizing full width tillage, fertilizer applications prior to tillage allows for incorporation. Table 5 shows Indiana data on the impact of UAN placement on corn yield. N placement had little impact on corn yields with conventional tillage, but injected UAN resulted in higher yields in no-till production than surface application, due to less N volatilization loss. If UAN solutions are applied to heavy crop residues, dribble application or banding may reduce contact with urease enzyme, slowing conversion of urea to ammonia, lengthening the time urea can remain on the surface before being incorporated by rainfall. Banding of ammonia fertilizers slows nitrification which reduces nitrate accumulation in the soils and minimizes nitrate leaching risk, particularly for early applications (Power et al., 2000).

Controlled release fertilizers with and without nitrification inhibitors are currently being evaluated. These new nitrogen sources could potentially improve nitrogen use efficiency and reduce nitrate leaching. Controlled release fertilizers include sulfur-coated and polymer-coated urea, among others. Stable-U is an experimental urea-calcium formulation designed to slowly release N, so that volatility and leaching losses may be reduced. Shoji et al., (2001) reported that there was no yield difference between traditional fertilizers and controlled release fertilizers when the latter where applied at a nitrogen equal to half of the traditional fertilizers.

Table 5. Effect of nitrogen (UAN) placement on corn yield, leaf N concentrations, and N concentrations in both plow and no-till production systems. Southeast Purdue Agricultural Center, 1987.

Nitrogen Placement	Yield (bu/a)	Ear leaf % N	Grain % N	Yield (bu/a)	Earl leaf % N	Grain % N
	Plowed			No-Till		
Broadcast - not incorp.	145	2.48	1.21	128	1.63	1.08
Broadcast – incorp.	153	2.34	1.23	-	-	-
Injected	149	2.44	1.29	156	2.13	1.04
LSD 0.05	NS	NS	NS	17	0.27	NS

Source: Mengel, D.B. 1989.

In ridged crops, such as ridge-till corn or potatoes, placing N fertilizers in a band in ridges make N less susceptible to leaching, and may improve N use efficiency (Hendrickson et al. 1978; Saffinga et al. 1976). Lowery et al. (1995) measured reduced leaching of nitrate when fertilizer was banded on the shoulders of ridges in ridge-till corn, compared to banding in furrows. The high osmotic potential produced in a fertilizer band inhibits nitrification in sandy soils, slowing conversion of ammonia to nitrate, reducing leaching risk (Hendrickson et al. 1978). Similarly, leaching of nitrate in furrow irrigated systems can be reduced by changing fertilizer placement. Placing N fertilizer in corn rows resulted in 12% more N uptake by the crop than when fertilizer was placed in furrows (Martin et al., 1995; Watts and Schepers, 1995; Benjamin et al. 1997).

Application Precision and Equipment Modifications

Nonuniform fertilizer or manure applications reduce yields where application is deficient and risk increased nutrient losses where application is excessive. Application equipment should be calibrated frequently to insure uniform application. Improvements in application equipment may allow more precise applications in the future.

Nonuniform application of anhydrous ammonia has received increased attention recently (Fee, 1997; Schrock et al., 1999). Even when the amount of ammonia per acre is correctly applied, amounts applied may vary significantly from outlet to outlet across the applicator. For example, in one calibration test of an anhydrous applicator, a manifold set for 140 lb/ac (157 kg/ha) application rate varied from + 40% over application to – 26% under application (Reichenberger 1994). At lower N application rates, variation was even greater, from + 132% to – 52%. Similar results were obtained by Boyd et al. (2000) who who found coefficients of variation ranging from 125 to 80%at 84 kg N/ha and from 10% to 66% at 168 kg N/ha. Meisinger and Delgado (2002) mentioned that the coefficient of variation among knives of conventional manifolds ranges from 10% to 70%. Another demonstration showed that applicators tested typically showed three to four times as much anhydrous ammonia exiting some knife outlets as others (Fee 1997). Such outlet to outlet variations are often invisible to equipment operators. Research is underway to design application equipment that more uniformly distributes fertilizer among outlets. For example, a Vertical-Dam manifold had less variability than conventional manifolds at a 56 kg N/ha (50 lb N/ac) application rate (Hanna et al. 1999; Boyd et al., 2000). At higher application rates there was little variability between the manifolds. Liquid fertilizer applicators usually are more precise and accurate than anhydrous ammonia applicators. Dinnes et al. (2002) applied nitrogen rates within 1% to 4% of the target with hydraulic flow control devices.

Other refinements in fertilizer applicators may reduce nitrate leaching losses. An N fertilizer injector that forms a locally compacted soil layer and a surface ridge or dome has been examined for effects on nitrogen losses and corn yields (Ressler et al. 1998). When the experimental injector was compared to a conventional injector, during seasons when rainfall was below average, neither nitrate leaching nor corn yield showed a response to fertilizer injection technique. In an above-average rainfall season, corn yields were higher with the experimental applicator and 23 lb/ac (26 kg/ha) more nitrate remained in the top 2.6 ft (0.8 m) of soil, compared to the conventional applicator. Point injectors of liquid nitrogen fertilizer have been shown to increase corn yield, total N uptake compared to surface broadcast (Randall et al., 1997) and reduced nitrate concentration in tile drains compared to knife injection (Dinnes et al., 2002)

Variable rate fertilizer applications promise to improve N use efficiency and reduce nitrate losses. Recent studies have documented that the optimal N rate for corn (Mamo et al., 2003; Scharf et al., 2005) and wheat (Fiez et al., 1994) vary spatially within fields. Historically, fields have been managed as a unit, with fertilizer rates uniform across the entire field. Due to variations in yield potential due to factors such as soil type (Oberle and Keeney, 1990), and variations in N availability due to factors such as soil organic matter (Clay et al., 1997; Soon and Malhi, 2005) or previous cropping or manure application differences, some areas of fields may receive too much N fertilizer, while other areas may receive too little. Using precise maps of soil variables and/or localized N needs determined from soil, tissue tests and remote sensing in season, N fertilizer can be applied at a variable rate to match the soil productivity potential or crop needs (Wiese et al. 2000; Redulla et al. 1996). Less leaching of nitrate below the root zone was documented in Washington potato production with variable rate N application (Whitley et al. 2000).

Irrigation Management

Lands that are irrigated in general are more prone to nitrate leaching due to coarse texture and intensive cropping. Application of irrigation water can cause percolation of water below the crop root zone, carrying nitrate with it particularly when water application exceeds crop requirement. Careful scheduling of irrigation based on soil moisture estimates and daily crop water needs can improve irrigation efficiency and reduce nitrate leaching. An irrigation scheduling system should consider soil water-holding capacity, evaporation, rainfall and previous irrigation, and crop growth stage to determine the timing and amount of irrigation water to be applied (Meisinger and Delgado, 2002).

The impact of irrigation scheduling and N rates were studied in corn grown on a sandy loam soil in Minnesota (Sexton et al. 1996). Sprinkler irrigation was applied to field capacity either at a fixed trigger deficit throughout the season, or at a variable trigger deficit based on crop growth stage. N was applied at various rates as urea, and yields and nitrate leaching measured. Applying N at rates at which corn yields were 95% of maximum reduced nitrate leaching losses by 35%. When a variable available water deficit was used to schedule irrigation rather than a fixed deficit schedule, nitrate leaching was reduced 46%. Schepers et al. (1991) surveyed 4000 commercial farms in Nebraska and found that improved N and water management reduced ground water nitrate concentration. Similar research being conducted in other regions should help to design irrigation scheduling techniques which produce profitable yields and protect water quality.

Sprinkler irrigation systems generally apply water more uniformly and in lower amounts than furrow irrigation systems, reducing nitrate leaching losses. A center-pivot reduced water application rate to 25-40 cm, compared to the rate applied with furrow irrigation that ranged from 100 to 140 cm and increased corn yield (Watts and Schepers, 1995). In these same studies, the authors found that nitrate leaching was also greatly reduced with center-pivot irrigation. In Nebraska, long-term nitrate concentrations were compared under four fields: 1) a conventional furrow irrigated corn field, 2) a surge-irrigated corn field which got 60% less water and 31% less N than the conventional field, 3) a center pivot irrigated corn field which got 66% less water and 37% less N than the conventional field, and 4) center pivot irrigated alfalfa (Spalding et al. 2001). Significantly less nitrate leaching was found below the center pivot corn field, leading to the conclusion, "The results demonstrate that the conversion from furrow to well-managed sprinkler irrigation would significantly benefit shallow ground water quality in the central Platte region and other corn growing areas of the western U.S.A." Additionally, the authors concluded that "surge irrigation was unable to satisfactorily limit NO_3-N leaching, negating any inherent water quality benefits of applying less water and N."

Sprinkler and drip irrigation, varying irrigation triggers, and N sources were compared in potato production in Minnesota (Waddell et al. 2000). Water percolation was generally higher from sprinkler irrigation than drip irrigation. When sprinkler irrigation was applied either at triggers of 70% available water (AW) remaining or 40%

AW remaining, the 70% AW treatment had the most nitrate leaching, followed by 40% AW and drip irrigation, which were equal. Splitting N applications 5 times vs. 3 times reduced nitrate leaching due to unforeseen rains. Sulfur coated urea reduced nitrate leaching. Turkey manure treatments had similar amounts of nitrate leaching as urea-N treatments.

While furrow irrigation may result in greater N losses than sprinkler irrigation, furrow-irrigation efficiency can be improved by adjusting set time, stream size, furrow length, watering every other row, or the use of surge valves. Running irrigation water through every other furrow and applying N fertilizer in the nonirrigated furrow reduced nitrate leaching losses in Nebraska (Martin et al. 1995). Similar results were found in Idaho (Lehrsch et al. 2001) where banding N in every other corn row and furrow irrigating non-fertilized rows maintained or increased N uptake by the crop and minimized residual nitrate in the soil after harvest. In furrow-irrigated onions in Oregon (Shock et al. 1997), adding wheat straw at 800 lb/ac (900 kg/ha) to furrows reduced runoff volume by 43%. Total N losses in the first 6 irrigations were 205 lb/ac (230 kg/ha) for unmulched and 29 lb/ac (33 kg/ha) for mulched treatments. Crop residue left on the soil surface by conservation tillage should produce a similar effect in slowing runoff and reducing erosion and nutrient losses.

Fertigation or applying N fertilizers through irrigation systems may facilitate supplying N when the crops demands are greatest. Multiple applications of relatively low N rates can be applied to correspond with periods of maximum crop uptake, reducing N available for leaching. When fertilizers are applied to the soil surface, one half inch or more of water from a sprinkler irrigation system can move N into the soil and minimize ammonia volatilization or immobilization.

Polyacrylamide (PAM) applications have been highly effective in reducing P losses with furrow irrigation (Lentz et al. 1998; Lentz et al. 2001). Although nitrate runoff losses with PAM treatment were not different from controls, the 90% reduction in sediment losses caused by PAM treatment would result in significant reductions in total N losses. PAM did not affect nitrate leaching.

Inhibitors

Nitrification and urease inhibitors can be used to increase N use efficiency under certain conditions. Nitrification inhibitors (nitrapyrin and dicyandiamide or DCD) are used with ammonium or ammonium-forming N fertilizers to improve N efficiency and limit losses of fertilizer N on soils where the potential for nitrate leaching or denitrification is high. These inhibitors temporarily suppress populations of *Nitrosomonas* and *Nitrosococcus* soil bacteria which are responsible for the conversion of ammonium to nitrate, thus keeping ammonium fertilizers in the non-leachable ammonium form. Maintaining N in the ammonium form also reduces losses from denitrification in water-logged soils. Nitrification inhibitors break down with time and have an effective period of 2 to 6 weeks, depending on moisture and temperature conditions (Ferguson et al. 1994).

Yield responses to nitrification inhibitors have varied with location and environmental conditions. Table 6 shows University of Wisconsin estimates of the probability of obtaining a corn yield increase with the use of nitrification inhibitors on various soils and with different application timings. While fall applications of N on sandy soils are discouraged in some areas, use of nitrification inhibitors may reduce N losses if N is fall applied to such soils. An 8-year study in Ohio showed that fall applied urea or anhydrous ammonia with a nitrification inhibitor increased corn yields with compared to the situation without the inhibitor (Stehouwer and Johnson, 1990). Randall et al. (2002) found that fall application with nitrification inhibitor had higher corn yield and nitrogen use efficiency than the without inhibitor. However, both studies showed that spring nitrogen application produced the highest yields emphasizing the importance of fertilizer application timing.

Table 6. Relative probability of increasing corn yields by using nitrification inhibitors.

Soil Type	Time of Nitrogen Application		
	Fall	**Spring Preplant**	**Spring Sidedress**
Sands & loamy sands	NR[1]	Good	Poor
Sandy loams & loams	Fair	Good	Poor
Silt loams & clay loams			
Well-drained	Fair	Fair	Poor
Somewhat poorly drained	Good	Good	Poor

[1]Fall applications not recommended on these soils

Source: Bundy, L.G. 1985. Corn fertilization. University of Wisconsin Coop. Extn. Serv. Bull. No. A3340. 8 p.

Rao (1996) evaluated nitrification inhibitors and urea placement for no-till winter wheat production in Oklahoma. Urea treated with DCD or nitrapyrin increased grain yield 7 to 31% above untreated urea. Warm temperatures and dry soil in the fall reduced the effectiveness of nitrification inhibitors when urea was surface applied. Placing treated urea 3 to 4 cm below the seed resulted in greatest reductions in nitrification, and highest yields.

Several studies have measured nitrate leaching losses with and without nitrification inhibitors. Owens (1987) measured nitrate leaching losses using monolith lysimeters containing a silt loam soil planted to corn in Ohio. Urea was applied at 300 lb N/ac (336 kg N/ha) either alone or with nitrapyrin. Over 6 years, the average annual nitrate-N loss was 104 and 143 lb/ac (117 and 160 kg/ha) for the nitrapyrin-treated and untreated urea, respectively.

Timmons (1984) measured nitrate leaching in sandy loam soil in laboratory columns and field lysimeters. In soil columns fertilized with 200 lb N/ac (224 kg N/ha), the addition of nitrapyrin reduced nitrate leaching losses by 46 and 27 lb/ac (51 and 30 kg/ha), respectively, for 0.5 and 1.5 in (12.7 and 38.1 mm) weekly water application levels. Annual nitrate-N leaching losses measured at the 3.9 ft (1.2 m) depth in field lysimeters cropped with corn over a 3-yr period averaged about 7% less for nitrapyrin-coated urea.

Urease inhibitors temporarily block the function of the urease enzyme, maintaining urea-based fertilizers in the non-volatile urea form. Use of these inhibitors may reduce volatilization losses when urea or UAN solution are applied to the soil surface in high residue, conservation tillage systems.

Crop Rotation

Nitrate losses in subsurface drainage are often highest for annual crops fertilized with N, especially corn. Legumes and other crops not needing supplemental N can utilize N remaining in the soil from previous crops. Logan et al. (1980) reported that nitrate losses in drainage water were greatest for N-fertilized corn, intermediate for soybeans or systems where other crops were grown in rotation, and lowest in alfalfa. Nitrate losses in tile drainage in Ontario were highest with continuous corn, intermediate with a corn-oat-alfalfa-alfalfa rotation, and lowest with continuous bluegrass (Bolton et al. 1970).

Alfalfa effectively reduces nitrate concentrations in the soil profile and has been recommended as a management alternative to remove nitrate from the soil below the rooting depth of most annual crops (Russelle and Hargrove 1989). In Nebraska, five years of alfalfa greatly reduced soil and water nitrate concentrations (Watts et al. 1997). Non-nodulating alfalfa varieties remove more nitrate from the subsoil than conventional nodulating varieties (Blumental and Russelle 1996).

Nitrate leaching losses are usually less for a corn-soybean rotation than for continuous corn, as long as proper N credits are taken for the soybeans (Rice et al. 1995; Kanwar et al. 1997; Albus and Knight 1998). Considerably more nitrate loss in tile drainage was measured for continuous corn than from corn-soybean rotation (Randall et al., 1993). Similarly, Katupitiya et al. (1997) observed greater nitrate leaching in furrow irrigated fields for continuous corn than for corn-soybean and Varvel et al. (1995) measured a 19% reduction in nitrate leaching for rotated versus continuous corn.

Randall et al. (1997) established 4 cropping systems in Minnesota to determine above ground biomass yields, N uptake, residual soil N, soil water content, and nitrate losses to subsurface drainage tiles as influenced by cropping system. Continuous corn, corn-soybean rotation, alfalfa, and Conservation Reserve Program or CRP (planted to a mixture of alfalfa, smooth bromegrass, orchard grass, and timothy) were compared for 6 years. Nitrate losses in subsurface drainage water from the continuous corn and corn-soybean systems were about 37X and 35X higher, respectively, than from alfalfa and CRP systems due primarily to greater season-long evapotranspiration resulting in less drainage and greater uptake and/or immobilization of N by the perennial crops.

Cover Crops

Nitrate is most subject to leaching loss following the harvest of annual crops up until the following crop begins utilizing N (Cambardella et al, 1999). In some studies 88 to 95% of nitrate leaching losses occurred during this period (Drury et al. 1996). Cover crops grown during the fallow period can scavenge N and other nutrients, preventing leaching. Meisinger et al. (1991) reviewed the effect of cover crops on nitrate leaching and concluded that cover crops can commonly reduce both the mass of N leached and the nitrate concentration in the leachate by 20 to 80%. Nnutrients are returned to the soil when the cover crop dies. Grasses and brassicas are particularly effective scavengers of soil residual nitrates, being rye (*Secale cereale* L.) the best adapted grass cover crop for the northern Corn Belt (Dinnes et al., 2002).

Substantial reductions in nitrate leaching have been reported with the use of cover crops. Martinez and Guiraud (1990) measured nitrate leaching with lysimeters in an irrigated corn-wheat rotation with and without a ryegrass cover crop and reported a 67% reduction in nitrate leaching with the cover crop. Also using lysimeter techniques, Bergström and Jokela (2001) reported that nitrate leaching was reduced by two-thirds in the first year and by more than 50% over a 2 yr period when a ryegrass cover crop was used in barley production. Lewan (1994) measured nitrate losses through drainage tiles in a sandy soil planted to barley. For 3 years when a perennial ryegrass cover crop was interseeded and plowed prior to spring barley planting, nitrate leaching was reduced 83%

compared to no cover crop. In the 4th year when the cover crop was plowed down and not replaced, nitrate leaching was greater than in plots with no cover crop history.

Rass et al. (2000) investigated the utility of rye cover crops in the production of inbred seed corn. At a 90 lb N/ac (101 kg N/ha) fertilizer rate, little nitrate was lost with or without a cover crop. However, at a 180 lb N/ac (202 kg N/ha) rate, the rye cover crop sequestered 41 to 50 lb/ac (46 to 56 kg/ha) of the excess fertilizer N.

Altered Drainage Tile Design

One third of the crop area in the Midwest is tile drained (Power et al., 2000). Because tile drainage can account for a large percentage of N losses through nitrate leaching (Jaynes et al., 1999), researchers have investigated means of either reusing tile effluent through controlled drainage or subirrigation systems or increasing denitrification of nitrate before or after drainage water enters tiles.

There are three strategies to reduce nitrates contamination with drainage control: increase denitrification, reduce the amount of drainage water, and decrease the water infiltration depth in the soil profile (Dinnes, et al., 2002).

Controlled drainage-subirrigation systems are constructed by placing water level control structures below tile outlets. By plugging lower drains from the structures, water levels can be raised, creating a pressure head which forces water back into the tile systems. Or drainage water may be pumped from holding ponds back into tile systems. In this way excess water can be used for subirrigation during dry periods and nitrate in drainage water utilized by the crop. Fisher et al. (1999) conducted a field-plot scale experiment in a corn-soybean production system to compare controlled drainage-subirrigation (water table maintained at 0.4 m) against subsurface drainage with no drainage control. They found that the controlled drainage treatment reduced nitrate concentration at the 30-75 cm depth by 46%, increased corn yield by 19% and soybean yield by 64% averaged over two years.

In Ontario, Drury et al. (1996) measured a 24% reduction in tile drainage volume using a controlled drainage system. Average annual nitrate loss was reduced by 43%. Using controlled drainage plus conservation tillage resulted in a 49% reduction in annual nitrate losses. In Italy (Borin et al. 2001), controlled drainage reduced nitrate losses by 46 to 63%. Fogiel and Belcher (1991) measured reductions in nitrate losses of 25 to 59% with controlled drainage. Using simulation models, Skaggs and Gilliam (1981) predicted that water table control could reduce nitrate movement to streams by up to 39%. Others have shown that nitrate leaching was reduced as water table depth decreased and as the time between fertilizer application and initiation of leaching increased (Jiang, et al., 1997) suggesting that denitrification is the main process responsible for nitrate removal. Nitrous oxide emissions increased when the water tables were less than 0.5 m deep, compared to deeper water tables (Jacinthe et al., 1999). Kalita and Kanwar (1993) observed significant reductions on nitrate concentration as depth to the water table decreased but also reported a negative effect of shallow water table on crop yield. This tile management practices are most effective when the environmental conditions are conducive to high denitrification rates. However, those same conditions negatively impact crop yield and limit the practical application of this strategy.

It may be possible to alter drainage tile construction to favor denitrification of nitrate. Research is currently underway in Iowa and Illinois to investigate burying organic materials as energy sources for denitrifying bacteria near tiles (McMahon 2000).

Installing tile deeper but keeping outlets at normal depths may also reduce nitrate in effluent by encouraging denitrification.

These water level control technologies are especially suitable for poorly drained soils with slopes less than 1% (Meisinger and Delgado, 2002). However, these practices increase management time and may be impractical in steeper slopes (Skaggs and Chescheir, 1999).

Conservation Buffers

Conservation buffers trap sediment containing ammonia and organic N. However, most nitrate is carried to streams by subsurface flow. In order to trap nitrate, buffers must somehow intercept or convert nitrate flowing beneath the buffer. Extensive study has shown that buffers are surprisingly effective in reducing nitrate losses to surface water (Jacobs and Gilliam, 1985; Verchot et al., 1997; Rickerl et al., 2000). Mechanisms of removal include uptake of nitrate by buffer roots, increased denitrification facilitated by organic matter energy sources placed in the subsoil by roots of buffer species, interactions with buffer vegetation and dilution (Dinnes et al., 2002).

Trees, shrubs, and other deep-rooted species are most efficient in intercepting deeper subsurface flow and placing organic carbon deep in the subsoil. Groffman et al. (1991) found that denitrification was more rapid in soil cores taken from grass buffers than from forest buffers. Conversely, Hubbard and Lowrance (1997) determined that a riparian forest was more efficient in assimilating nitrates than a grass buffer. Nitrate removal by riparian buffers has been documented in many settings (Cooper 1990; Simmons et al. 1992; Lowrance 1992; Jordan et al. 1993).

In North Carolina, Spruill (2000) conducted a statistical evaluation of groundwater discharging to a stream. Nitrate was 95% lower in buffer areas compared with nonbuffer areas. Dilution was estimated to account for 30 to 35% of the difference, with reduction and/or denitrification accounting for 65 to 70% of the difference.

Addy et al. (1999) demonstrated the importance of carbon enrichment of subsoil on nitrate removal by buffers. Undisturbed 15.7 in (40 cm) deep soil cores were taken from beneath forest and mowed buffers and dosed with radiolabeled nitrate. Nitrate removal rates were highly correlated to carbon-enriched patches of organic matter. There was no difference between the forest or mowed cores. Groffman et al. (1991) found that adding carbon as glucose to soil cores from grass and forest buffers increased denitrification. Some evidence suggests that the residence time of water is a key factor determining the effectiveness of denitrification. Areas where water flow is slow are more effective than areas where water moves rapidly (Snyder at al., 1998; Lowrance et al., 2000).

Constructed Wetlands

While conservation buffers are effective in reducing nitrate losses to surface water, in many agricultural areas, drainage tiles carry a significant amount of subsurface water directly to streams, preventing any interaction with buffers. However, it is possible to process tile effluent in wetlands prior to its release to streams and reduce nitrate concentration (Crumpton et al., 1995; Romero et al., 1999). Wetlands reduce nitrate concentrations by favoring denitrification.

Crumpton and Baker (1993) measured the ability of wetlands to denitrify nitrate and used a model to determine the size of wetland required to reduce tile inflow of a given nitrate concentration and volume to meet drinking water standards. Tile discharges containing up to 24 ppm nitrate-N from a 247 acre (100 ha) drainage field could be cleaned up to less than the 10 ppm nitrate-N standard by passing through a 2.47 acre (1 ha) wetland.

Kovacic et al. (2000) constructed 3 wetlands of 0.7 to 2.0 ac (0.3 to 0.8 ha) to intercept tile drainage. The drainage area to wetland surface area ratios were from 17 to 32 for the 3 wetlands. Over a 3-yr period, 37% of N input was removed by the wetlands, with the nitrate concentration reduced by 28%. When the wetlands were coupled with a 50 ft (15.3 m) vegetated buffer between the wetland and river, an additional 9% of tile nitrate was removed.

Research is underway at a number of locations to further document the ability of constructed wetlands to reduce nitrate loads to surface water and to help determine optimal sizes of wetlands. Water residence time in the wetland seems to be a key factor controlling its effectiveness to remove nitrates from water (Crumpton et al., 1995).

Pest Management

Because N fertilizers rates are based on a projected crop yield, if yields are depressed due to an unanticipated problem such as an insect or disease outbreak, N may be excessive and risk increased losses to water. Thus, maintaining a healthy crop by managing insects and diseases is important to N management. Insects and disease management involves many practices including planting resistant crop varieties and appropriate use of insecticides and fungicides.

New biotech crops may aid in pest control and N management. The European corn borer causes premature death of corn plants by tunneling in stalks and increasing associated stalk rots. Bt corn varieties have been highly effective in controlling corn borers and preventing early death of corn plants (Ostlie et al. 1997). Keeping the corn plant green and healthy throughout the growing season allows it to take up N provided in anticipation of normal yields, preventing loss of excess N through leaching.

Breeding Crops for Efficient N Uptake

The ability of crop varieties to extract and utilize nutrients from the soil may vary significantly (Bertin and Gallais, 2000). Research in Indiana (Tsai et al. 1984) showed that the yield potential of different corn hybrids may be related to the N response characteristics of the hybrid. More recently, O'Neill et al. (2004) found a significant difference among corn hybrids in the yield they achieved without N fertilizer and in their response to N fertilization. Certain hybrids adapted to low fertility might require little N fertilizer and take up most of N needs early in the season, while high fertility hybrids might respond to higher N rates over a longer period. Huffman (1986) suggested that low fertility hybrids could be managed for optimum N use efficiency on droughty, low organic matter, sandy soils. Improved N efficiency through corn hybrid selection has not been consistent (Bundy and Carter 1988). However, selective breeding for N use efficiency may produce corn and other crops that more efficiently take up N,, are more N stress tolerant and have a high yield potential (O'Neill et al., 2004). Molecular markers have been used to identify candidate genes to breed for nitrogen use efficiency (Gallais and Hirel, 2004).

BMPs to Reduce Phosphorus Losses

Soil Conservation Practices

Total P and available P runoff losses of are highly correlated to sediment losses or erosion (Baker and Laflen 1983; Mueller et al. 1984; Cox and Hendricks 2000; Aase et al. 2001; Bundy et al. 2001; Uusitalo et al. 2001; Daverede et al. 2004). Soil conservation practices such as conservation tillage, contour tillage and planting, and terraces have shown to be effective in reducing P runoff losses. Tolbert et al. (1999) investigated the impact of added corn residue on nutrient runoff from a clay soil in Texas. Adding residue in a chisel-till system reduced total P runoff nearly 7-fold, compared to no added residue. In another study in Wisconsin, Grande et al. (2005) observed that total and dissolved P loads in runoff from rainfall-simulated plots were inversely related to percent residue cover. However, residue cover did not affect total and dissolved P concentrations. This is because concentrations of dissolved P and total P depend on other management factors such as fertilizer or manure placement and soil P extractable levels. Since particulate P usually represents about 80% or more of the total P load of conventional row crop runoff (Logan 1987; Uusitalo et al. 2001; Daverede et al. 2004), minimizing erosion with conservation practices will necessarily bring about reductions in total P in runoff.

Phosphorus Rates

Basing P application rates on soil test results has been highly effective from an agronomic standpoint, with none of the uncertainties encountered with N testing. Agronomic soil tests extract all or a proportionate amount of plant-available P. When soil test data is correlated to crop responses to P additions under varying soils and conditions, recommendations can be developed for fertilizer or manure application rates for optimal crop production. While P rates have been selected based on economics in the past, water quality considerations may be more important in the future. Phosphorus rates considered optimum for crop production may be considered excessive from an environmental standpoint in some vulnerable settings. Because runoff of P in surface water is highly correlated to soil test values (Pote et al., 1996, 1999; Hooda et al., 2000; Sauer et al., 2000; Sharpley et al. 1996; McDowell and Sharpley 2001; Daverede et al., 2003; Klatt et al., 2003), determining environmentally sound P application rates will be critical to protecting water quality.

Soil Testing Procedures

Agronomic soil tests are interpreted to determine, based on the amount of P that can be chemically extracted from the soil, the likelihood that crop yield will be improved enough by the application of P to equal the costs of applied P. Critical P values are thus established for various crops and soils, identifying P values at which crops are likely to respond to P additions. Because soil properties such as pH affect the efficiency of specific laboratory P tests, different tests are used in different regions and for differing soils within regions. Common agronomic P soil tests in use include Mehlich 1, Bray P1, Mehlich 3, Morgan and Modified Morgan, Olsen, and AB-DTPA (Sims et al. 1998).

Because agronomic soil tests have been calibrated for agronomic and economic purposes, without specific environmental calibration and interpretation they are not useful in determining environmentally critical P values. Traditional soil tests may prove useful for determining such environmental values, with appropriate calibration. Or new testing procedures may be needed to quantify the likelihood that soil P will cause environmental problems.

Because of widespread use of traditional agronomic soil tests and the extensive data base that exists, researchers have tried to correlate these test results to dissolved P and bioavailable P tests used for environmental purposes. Fortunately, many studies have shown that traditional soil tests are positively correlated with dissolved P and/or bioavailable P in surface runoff and subsurface drainage (Wolf et al. 1985; Pote et al. 1996; Sharpley et al. 1996; Heckrath et al. 1995; Smith et al. 1995; Hooda et al., 2000; Sauer et al., 2000; Sims et al., 2000; McDowell and Sharpley 2001; Daverede et al., 2003; Klatt et al., 2003).

Even if traditional soil tests are used to determine critical environmental values, soil sampling techniques may need to be modified. Agronomic soil tests are usually taken 6 to 8 in (15 to 20 cm) deep, since this is the depth to which most tillage implements operate and where most crop roots grow. However, P in surface runoff and erosion is affected most by a very shallow surface soil layer one or two inches (2 to 5 cm) deep (Pote et al. 1996). Thus, shallow soil tests may be needed to most accurately predict P runoff, especially if P is stratified and is more concentrated near the soil surface, as in no-till fields that have had regular applications of manure or fertilizer (Kingery et al., 1994; Sims et al., 1998).

Sampling both shallow and at normal sampling depths could determine the feasibility of reducing soil P concentrations in shallow, runoff-prone depths by deep tillage (Moore et al. 1998). Sharpley (2003) studied the effect of plowing on the homogeneization of P- stratified soils and found that total and dissolved P concentration in runoff was reduced significantly under a simulated rainfall 20 weeks after tillage and grass planting.

Alternative tests may be useful in predicting environmental P losses. The iron oxide strip or Pi soil test uses a strip of filter paper coated with Fe-oxide (Sharpley 1993; Chardon et al. 1996). When placed with a soil sample and a dilute salt solution, the strip acts as an "infinite sink" for the P that can be desorbed from the soil and thus measures the potential of a soil to continue releasing P during a runoff or leaching event. Water-extractable P has also been used successfully as an environmental P test. As distilled water is very similar to rainfall, it might be expected to simulate the rapid release of P to runoff water better than the stronger chemical extractants used in agronomic P tests (Pote et al. 1996). Vadas et al. (2005) investigated 17 publications to compare extraction coefficients (slope between dissolved P and soil extractable P) to model dissolved P release from soil to runoff. They found that a single extraction coefficient (2.0 for Mehlich-3P and 11.2 for water-extractable P data, and a split-line relationship for P sorption saturation data) could be used to predict dissolved P release from soil to runoff water for most edaphic, hydrologic, or management conditions.

Environmental P Thresholds

Several states have developed environmental threshold values for soil P tests, using the same soil tests used for agronomic purposes (Table 7). Environmental threshold levels range from 2 times (Michigan) to 4 times (Texas) the agronomic thresholds. As data is gathered for more soils and locations, additional states may develop environmental thresholds, and existing thresholds can be expected to be refined. Daniel et al. (1998) have reviewed environmental P thresholds. The National Phosphorus Research Project, a nation-wide cooperative project between USDA-ARS, USDA-NRCS, USEPA, and land grant universities, is investigating the relationship between soil P and runoff P and will provide information helpful in setting environmental P thresholds. In April 1999, the USDA and NRCS issued a national policy statement on nutrient management. This statement included guidelines on nutrient management, including organic amendments. Three P risk assessment tools were suggested by these federal agencies: agronomic soil test P interpretation classes, environmental soil P threshold limits, or a P index (explained in the Phosphorus Index section). According to Mallarino et al. (2002), a particular soil test P level or P application rate may cause different P losses depending on the particular field characteristics, including soil properties, landforms and management. Therefore, the P index is recognized as being a more comprehensive way of assessing the potential risk of P runoff from a field, since it takes into account many factors that affect P runoff, including erosion, distance to a water body, and soil test level.

Application Timing

While fall application of N is often discouraged due to risk of nitrate leaching, P can be applied in the fall if protected from runoff. Surface application in the fall can result in excessive runoff losses in some settings. Application to frozen soils especially risks P runoff during snowmelt (Klatt et al., 2003) or thawing. Meals (1996) measured increases in P export of up to 1500% with winter spreading of manure onto corn fields, with up to 15% of the applied P lost in runoff.

Fall tillage can incorporate applied P fertilizers and manure, reducing runoff. Baker and Laflen (1983) surface-applied fertilizer and incorporated it in some treatments after soybean harvest. Total P losses in erosion and runoff were higher for surface application than any of the incorporation techniques (chisel-plow or disk). P losses were lower for chisel-plow incorporated fertilizer than disk incorporation. While fall tillage may be discouraged in sloping fields due to erosion concerns, if tillage is conducted, fertilizer and manure application prior to tillage can reduce the risk of P runoff.

Application of fertilizer to wet soils increases runoff risk. Tolbert et al. (1999) measured P runoff losses from a clay soil in Texas under dry (350 g/kg moisture) versus wet (500 g/kg moisture) conditions at application. Greater P losses under rainfall simulation techniques occurred with application to wet soils, regardless of tillage and residue levels. Application to wet Bermuda grass sod resulted in a loss of 41% of applied P. Thus, fertilizer and manure applications to wet soils should be avoided.

Fertilizer and Manure Placement

The ability of incorporation or injection of fertilizers and manure to reduce P runoff has already been discussed in the Conservation Tillage section. Manure management will be discussed in detail in the Livestock and Manure Management section.

Starter fertilizer. Many agronomists have noted that crop responses to starter fertilizer placed near crop seeds is more common with conservation tillage than conventional tillage. For example, in Indiana in 11 experiments where starter fertilizer treatments were used in both no-till and conventional tillage for corn, starter fertilizer responses were obtained only in one case in conventional tillage, but in eight of the 11 experiments with no-till (Mengel 1989). Crop responses to starter fertilizer may be due to N, P, or K, depending on region and soil fertility.

Starter applications of P can usually supply all of the maintenance P requirements for row crops. Thus, if soil test P is in an optimum range, all maintenance P can be applied as a subsurface band at planting, reducing runoff. Phosphorus may be incorporated with tillage if large, corrective fertilizer P applications are needed,. Banding of P, whether as starter or as corrective applications, may improve the availability of P due to higher concentrations in the band. Schwab et al. (2006) studied the effects of tillage (moldboard plowing, disking followed by cultivation, and no-tillage) and fertilizer P application (broadcast, banded and a control with no P) on P uptake and grain yield for P-stratified soils. They recommended banding fertilizer when soils are highly stratified and have medium or low available P under 15 cm. This practice increased yield and P uptake of corn and sorghum, however it did not impact soybean yields significantly. Tillage effects on grain yield were inconsistent.

Variable rate application. P soil test values often vary considerably across fields due to soil type changes and varying histories of fertilizer and manure applications. Intensive grid soil sampling combined with variable rate application of P fertilizer can reduce fertilizer inputs and avoid over-fertilizing high-testing areas of fields. In Iowa, Mallarino et al. (1999) compared uniform and variable P fertilization on two corn and two soybean fields, using GPS techniques and grid soil sampling. Variable-rate fertilization reduced considerably the amount of P fertilizer applied in two of the four fields and increased yield in one field.

Irrigation Management

Runoff from furrow irrigation can carry P to surface water. Westermann et al. (2001) found that soluble and bioavailable P in furrow irrigation runoff were correlated to agronomic soil test P levels, while total P runoff was correlated to erosion and sediment losses. Surface crop residues in conservation tillage systems can reduce P losses. When wheat straw was added at 800 lb/ac (900 kg/ha) to furrow irrigated fields, P lost in 6 irrigations was 16 lb/ac (18 kg/ha) for the mulched plots compared to 192 lb/ac (215 kg/ha) for unmulched plots (Shock et al. 1997).

High molecular weight, anionic polyacrylamide (PAM) interacts with soil to reduce erosion losses with surface irrigation. Lentz et al. (1998) investigated the impact of two PAM treatments on P losses with furrow irrigation on a silt loam soil. PAM was applied at 10 mg/L only during furrow advance (stopped when runoff began) or applied at 1 mg/L continuously during irrigation. Both treatments were effective in reducing soil and P loss. Ortho and total P losses were 5 to 7 times lower with PAM. The authors concluded: “PAM is effective, convenient, and economical, and greatly reduces P and organic material (COD) losses from surface-irrigated fields.” A thorough review on PAM chemistry and product synthesis and characteristics, as well as its effect and fate in soil and plant systems can be found in Barvenik (1994), where the author concluded that PAM is safe if used according to specified directions.

Conservation Buffers

Because conservation buffers effectively trap eroded sediment, these buffers also trap sediment-adsorbed P. Under controlled conditions, buffers have removed a high percentage of total P in runoff. Dillaha et al. (1989) used rainfall simulation techniques to measure the effectiveness of 15 ft and 30 ft (4.6 and 9.1 m) vegetated strips below a 60 ft (18.3 m) plot of bare ground to which fertilizer had been applied. The 15 and 30 ft buffers trapped an average of 70 and 84% of incoming suspended solids, and 61 and 79% of incoming P, respectively.

Magette et al. (1989) also used rainfall simulation techniques to study nutrient trapping by 15 and 30 ft (4.6 and 9.1 m) vegetated strips below bare plots 72 feet (22 m) long. They found that P trapping efficiencies were variable, and decreased as the number of runoff events increased. Averaged over all tests, 46 and 27% of total P was trapped by 15 ft and 30 ft buffers, respectively.

Other controlled studies with similar sized buffers have measured total P trapping efficiencies of 44 to 96% (Thompson et al. 1978; Young et al. 1980; Doyle et al. 1977). While buffers are most effective in trapping sediment-adsorbed P, soluble P can also be significantly reduced, often due to increases in water infiltration in buffers. In France, grassed buffers trapped 22 to 89% of soluble P in runoff (Patty et al. 1997). Uusi-Kämppä et al. (2000) reviewed buffer studies conducted in Finland, Norway, Sweden, and Denmark. Buffers decreased total P loads from agricultural land runoff by 27 to 97%. Increasing buffer width improved retention, but upper parts of buffers were most efficient in trapping P due to the importance of sedimentation.

Castelle et al. (1994) reviewed buffer studies to determine appropriate buffer sizes. Depending on local conditions and intended buffer functions, a range of buffer widths from 10 to 650 ft (3 to 200 m) was found to be effective, with at least 50 ft (15 m) necessary to protect surface water under most conditions.

Perhaps more important than width of buffers is their configuration and maintenance. In order for buffers to efficiently trap sediment and nutrients, sheet flow needs to occur across the buffer. Concentrated or deep flow across the buffer can bypass most filtering effects. Dillaha et al. (1988) analyzed 33 existing buffers in Virginia for sediment trapping efficiency. They found that sediment trapping was often poor because of either concentrated flow where topography was hilly or sediment that accumulated in the buffer, causing runoff to flow parallel to the buffer until a low point was reached where concentrated flow occurred. Buffers should be constructed to cause as much sheet flow as possible. Stiff-stemmed grass hedges, berms, or other devices may be useful in directing runoff across buffers. As sediment accumulates in buffers, changing their profile, sediment will need to be removed or buffers reshaped.

Conservation buffers can be planted to perennial grasses, legumes, woody plants, or a combination. Species should be selected based on adaption to local conditions and desired benefits of buffers (water quality, wildlife habitat, etc.).

Cover Crops

Cover crops effectively reduce erosion during the period of time between when annual crops are harvested and the following crop covers the ground, and thus reduce runoff of sediment-adsorbed P. Cover crops may also take up available P, reducing chances for leaching losses of soluble P. Cover crops improve water infiltration, which reduces soil runoff. Singer and Kasper (2006) indicated that reduction of total phosphorus could range from 54 percent to 94 percent in research with cover crops. Reduction of phosphorus research was variable and reductions were not always achieved.

Constructed Wetlands

While constructed wetlands are most often installed to encourage denitrification of nitrate from drainage water, wetlands also have the capability of retaining P carried either in surface runoff or subsurface flow. Casey and Klaine (2001) investigated the impact of a riparian wetland receiving runoff from a golf course, finding that attenuation of P averaged 74%. The calculated soil P concentration that would yield an equilibrium aqueous P concentration of 50 ppb was found to be 100 times greater than the wetland soil P concentration, meaning that the wetland could retain a large additional mass of P without increasing dissolved P above EPA's recommended limit (Casey et al. 2001). In a review of Scandinavian studies, Uusi-Kämppä et al. (2000) report that wetlands reduced P loads an average 41%.

Little P has been removed by wetlands in some studies, especially when loads are mainly soluble P. Kovacic et al. (2000) constructed three wetlands to process drainage tile effluent (drainage area to wetland surface area ratios of 25, 17, and 32). In three years of monitoring, total P removal averaged only 2% and was highly variable between years.

Phosphorus Index

Identification of critical areas subject to the most P loss to water will allow the targeting of BMPs and reduce potential hardships for farmers, should required management changes reduce profitability. Areas most vulnerable to P loss are often small, well defined areas near stream channels and covering < 20% of watersheds (Heathwaite et al. 2000). Pionke et al. (1987) report that in some settings 90% of annual algal-available P export from watersheds come from only 10% of land area during a few large storms. In contrast, larger areas contribute nitrate and generally occur on the upper boundaries of the watershed where freely draining soils and high manure and fertilizer N applications are made.

In 1990, the USDA Natural Resources Conservation Source (NRCS) formed a group of scientists from Universities, Cooperative Extension, and USDA Agricultural Research Service to develop a P indexing procedure that could identify soils, landforms, and management practices with the potential to lose excessive P to surface water. A field-based planning tool, called the Phosphorus Index, was developed which integrated, through a multi-parameter matrix, the soil properties, hydrology, and agricultural management practices within a defined geographic area, to assess the risk of P movement from soil to water (Lemunyon and Gilbert 1993).

The P Index in a modified version (Sharply et al. 1999) uses nine characteristics to obtain an overall risk rating for a site. Each characteristic is assigned an interpretive rating with a corresponding numerical value: LOW (1), MEDIUM (2), HIGH (4), or VERY HIGH (8), based on the relationship between the characteristic and the potential for P loss from a site. Suggested ranges appropriate to each rating for a site characteristic are assigned. A weighting factor reflecting the relative importance of each site characteristic to P loss is also assigned. These weighting factors were based on the professional judgment of the scientists that developed the P Index. Individual states were encouraged to modify these site characteristics and their weighting factors, based on local soil properties and hydrologic conditions as field research is conducted to refine the index. Adding the weighted loss rating values for the nine site characteristics yields a numerical P index, which is related to general vulnerability to P loss. Specific management options are then suggested for use in each vulnerability category. Many states have developed modified P indexes that combine soil, P transport, and management factors in ways that respond to each particular situation. Benning and Wortmann (2005) compared these factor effects in different field scenarios for five P indexes from four Midwestern states, and found that the impact of these factors to runoff P risk assessment scores varied considerably between P indexes. For example, soil P levels increases from 30 to 90 mg kg^{-1} increased scores from 0 to 300% depending on the P index considered.

Livestock and Manure Management

Organic sources, such as manure, sludge, and compost can supply all or part of crop nutrient needs. Besides providing N, P, and K, secondary nutrients and micronutrients are also supplied. Organic amendments add organic matter to the soil (Anderson and Peterson, 1973; Hountin et al., 1997), which improves soil structure and soil quality, increases water infiltration and water holding capacity, and reduces erosion potential. Long-term manure applications can increase total N levels in soils (Muñoz et al., 2003), accompanied by increases in potentially mineralizable N levels (Ndayegamiye and Coté, 1989), decreasing the response of crops to N fertilization (Whitmore and Schroder, 1996; Mulvaney et al. 2001). While application of organic nutrient sources provides many benefits, significant risks to water quality occur when these amendments are applied at rates in excess of crop needs or under conditions that favor runoff or leaching.

To be able to take credit for nutrients in manure, nutrient content must be determined. Table 11 shows representative ranges of values for nutrients in manure, sludge, and whey. However, the nutrient content of these sources can vary greatly depending on many factors such as storage and handling procedures, climate, and management. Manure should be sampled and analyzed in order to accurately determine amounts of nutrients provided (Rieck and Miller 1996).

One problem in utilizing manure as a nutrient source is that N and P amounts may not be present in the ratio used by crops. If manure is applied based on crop N needs, excess P will likely be applied. Manure has often been applied based on crop N needs in the past where inadequate land was available to assign manure produced by concentrated livestock feeding operations. This practice sometimes increased soil P to excessive levels, risking pollution of nearby water sources.

Because organic N in manure must be mineralized before it can be utilized by crops, N is slowly released, with some N becoming available in years following application. Failure to consider this N carryover can result in excessive application of N. About 30 to 40% of the total N in dry beef cattle and dairy manure is available for crops the year of application with 10% available the second year and 5% the third year. All of the N in swine manure from liquid handling systems is available the first year and 65% of the total N in poultry manure is available the first year of application (Killorn and Lorimer 1999). These estimates of availability do not consider possible ammonia volatilization at the time of application. P is present in both organic and inorganic forms in manure. From about 60 to 100% of the total P in manure will be available to crops the year of application, depending on the type of manure (excreted from ruminant or monogastric animals), the soil adsorption characteristics and P mineralization (Killorn and Lorimer 1999).

Table 11. Representative values for nutrients in manure, sludge, and whey, as applied.

Solid Manure		**Total N**	**P_2O_5[1]**	**K_2O[1]**
Species	**% dry matter**	**lb/ton**		
Dairy cattle	18-22	6-17	4-9	2-15
Beef cattle	15-50	11-21	7-18	10-26
Swine	18	8-10	6-9	7-9
Poultry	22-76	20-68	16-64	12-45
Sheep	28	14-18	9-11	25-26
Horse	46	14	4	14
Liquid Manure		**Total N**	**P_2O_5[1]**	**K_2O[1]**
Species	**% dry matter**	**lb/1000 gal**		
Dairy cattle	1-8	4-32	4-18	5-30
Beef cattle	1-11	4-40	9-27	5-34
Veal calf	3	24	25	51
Swine	1-4	4-36	2-27	4-22
Poultry	13	69-80	36-69	33-96
Digested Sludge		**Total N**	**P_2O_5[1]**	**K_2O[1]**
		lb/1000 gal		
		20	12	1
Whey		**Total N**	**P_2O_5[1]**	**K_2O[1]**
		lb/1000 gal		
		12	9	18

[1]Convert values for P_2O_5 and K_2O to P and K by multiplying by 0.43 and 0.83, respectively.

Sources: Midwest Plan Service. 1985. *Livestock Waste Facilities Handbook*. Iowa State University, Ames, IA. Klausner, S. 1995. *Nutrient Management: Crop Production and Water Quality*. 95CUWFP1, Cornell University, Ithaca, NY. University of Wisconsin-Extension and Wisconsin Dept. of Agriculture, Trade, and Consumer Protection. 1989. *Nutrient and Pesticide Best Management Practices for Wisconsin Farms*. WDATCP Technical Bulletin ARM-, Madison, WI. University of Vermont. 1996. *Agricultural Testing Laboratory – Manure Analysis Averages, 1992-1996*. Dept. of Plant & Soil Science, University of Vermont, Burlington, VT. As compiled by North Carolina State Univ. in Guidance on Controlling Agricultural Sources of Nonpoint Source Pollution.

Manure Application

Field selection. Because manure has often been viewed as a waste product for disposal in the past, manure has sometimes been applied at excessive rates to fields near manure storage facilities. Excessive P levels in soil in high risk fields may preclude further manure applications until P levels drop. Fields with soils testing low in P benefit most from manure applications. Application of manure at rates based on crop N needs to such fields allows P in excess of crop needs to build P soil test levels to agronomically optimal levels.

Centralized lists of manure generators and farmers having land which would benefit from manure may facilitate the application of manure at environmentally and agronomically sound rates. In Delaware, a local poultry trade association established a manure bank network that puts manure-needy farmers in touch with manure-rich poultry growers (Sharpley et al. 1999). Other innovations could help to distribute manure to appropriate sites. It has been suggested that grain or feed trucks and railcars could transport dry manure back to grain source areas instead of returning empty (Collins et al.

1988). Such a program would have to address sanitary concerns. In Iowa, a data base of soil properties including organic matter, pH, available P, total P, and K is being developed to inventory soil nutrient status and facilitate the planning and placement of livestock facilities and determine where manure can be most judiciously used in crop production (Iowa Nutrient Management Task Force 2000).

Factors besides soil P content need to be considered in selecting fields for manure application. Besides nutrients, manure contains other substances which may be water quality concerns if runoff occurs. Organic solids reaching surface water may cause oxygen concentrations in water to drop as they decompose. These solids also provide a long-term nutrient source to aquatic systems. Bacteria, viruses, and other microorganisms may also enter water. Potential manure application fields should be assessed for the risk of runoff or leaching. Manure may also contain heavy metals, hormones, and livestock antibiotics and drugs. State or local regulations may also exist requiring untreated setbacks from streams, wells, sinkholes, surface tile inlets, residences, etc.

Application of manure to soils less than 10 in (25 cm) thick over fractured bedrock risks contamination of ground water (Madison et al. 1986). In karst areas, where sinkholes provide a direct conduit for surface runoff to ground water, incorporation of manure reduces ground water contamination potential. Risks to surface water are greater when manure is applied to sloping fields near streams and other water bodies. Application of manure to floodplains during high risk periods for flooding is not advisable unless manure is incorporated.

Application method and timing. Surface applied manure is subject to rapid loss of N through ammonia volatilization, especially under warm, dry and windy conditions. In contrast, incorporation reduces volatility losses, saving N for crop needs (Chase et al., 1991; Dosch and Gutser, 1996; Chadwick et al., 2001). Mooleki et al. (2002) observed higher N use efficiencies and more available N when swine manure was injected, compared to surface application followed by tillage.

Surface application also leaves manure vulnerable to runoff which could carry P (as well as organic enrichment and pathogens) to surface water. Surface application during winter and early spring months is a particular concern, as spring thaws or rains can cause significant runoff. In Vermont, winter spreading of manure onto corn fields increased P losses by up to 1500%, with up to 15% of the applied P lost in runoff (Meals 1996). If manure applications must be made to frozen soils, they should preferably be limited to soils with slopes of less than 6% (Madison et al., 1986).

P losses from manure applications are not necessarily greater than losses from other P sources. Tabbara (2003) found higher concentrations and load of all P forms from plots receiving broadcast P fertilizer compared to plots receiving surface applied liquid swine manure. Similarly, Withers et al. (2001) found that dissolved P concentrations in runoff from surface-applied triplesuperphosphate (TSP) almost doubled those from surface-applied cattle manure. When incorporated in the fall, dissolved P runoff concentrations were less than 0.5 mg/L for both treatments. In Minnesota, when N was applied as turkey manure or urea to corn grown on an irrigated sandy soil, less nitrate leaching occurred from turkey manure, while yields were equal for both treatments (Sexton et al. 1996).

Beneficial effects of manure on soil structure may sometimes partially offset increases in nutrient loss with surface application. Wendt and Carey (1980) found that surface application of manure to corn and alfalfa increased P concentrations in runoff, but did not increase P losses, due to increased infiltration where manure was applied. Bundy et al. (2001) found that spring applied dairy manure reduced total P runoff losses compared to no manure in chisel plow, shallow tillage, and no-till systems, apparently due to increases in infiltration.

Total P in runoff is highly dependent on the sediment concentration in runoff. Cox and Hendricks (2000) reported more than three-fold increase in TP concentration in runoff from conventionally tilled compared to no-till soils for a wide range of soil P levels on 2-6% slopes. Daverede et al. (2004) observed that Bray P1 soil extraction values and sediment concentrations and loads in runoff from incorporated swine manure and TSP were significantly related to total P and algal-available P concentration and loads. However, only Bray P1 soil extraction value influenced DRP concentration and load in runoff from injected manure and chisel-plowed TSP and control, and the relationship was linear. Bundy et al. (2001) found that incorporated manure generally lowered concentrations of dissolved P in runoff, but increased total P concentrations and loads due to increased erosion. Mueller et al. (1984) measured runoff losses of total P, algal available P, and dissolved P with and without manure applied prior to tillage in conventional tillage, chisel plow, and no-till systems (manure surface applied in no-till). Total P and algal available P losses were related to erosion, with unmanured no-till and chisel plow plots lower than conventional tillage. There was little difference in dissolved P losses between unmanured tillage treatments. However, unincorporated manure applications to the no-till increased soluble and algal available P losses. Total P losses with unincorporated manure on no-till were similar to losses where manure was incorporated in conventional tillage. When applying manure and compost on wheat residue plots with a 6% slope, Eghball and Gilley (1999) found that runoff DRP and AAP concentrations were greater for no-till than disked treatments during two consecutive simulated rainfall events. In contrast, concentrations of TP and PP were greater for the disked treatments compared to the no-till plots.

Incorporation or injection of manure is recommended where possible for both environmental and economic reasons. Tillage systems which incorporate manure while leaving significant amounts of crop residue on the soil surface should help to reduce both erosion and nutrient runoff. Application of manure ahead of tillage planned for seedbed preparation avoids the need for extra tillage solely to incorporate manure.

Manure management in no-till systems can be a challenge. Liquid manures may be injected with minimal disturbance of protective surface crop residue, but dry manures must be surface applied, risking increased runoff of dissolved P. Using limited tillage to incorporate dry manure in one year over a several year period may allow most of the benefits of no-till production to be realized, while utilizing manure in an environmentally sustainable manner. Application of liquid (< 2% solids) manure to the soil surface during periods of low intensity rainfall may not result in significant runoff.

Manure amendments. Alum $[Al_2(SO_4)_3\_16H_2O]$ addition to poultry litter has shown to produce significant reductions in P solubility of poultry litter (Moore and Miller, 1994; Sims and Luka-McCafferty, 2002; Miles et al. 2003 DeLaune et al., 2004), as well as reducing P concentrations in surface runoff (Shreve et al., 1995; DeLaune et al., 2004; DeLaune et al. (2006) The dissolved P of surface runoff from fescue treated with alum-amended poultry litter (11 ppm) was much lower than that from fescue treated with un-amended litter (83 ppm) (Schreve et al. 1995). DeLaune et al. (2006) found up to 84% reductions of dissolved P concentrations in runoff water under simulated rainfall when poultry litter was composted and treated with alum. In another study, Smith et al. (2001) observed that adding alum to swine manure at 430 mg/L reduced soluble reactive P concentrations in runoff by 84% when applied to tall fescue plots. Besides reducing soluble P losses, such amendments could help change the N:P ratio of manure by preserving N, so that manure applications based on crop N needs would reduce excess P added. When comparing alum-treated poultry litter, normal poultry litter and triple superphosphate as fertilizer sources for corn in Virginia, Warren et al. (2006) found that alum-treated poultry litter not only reduced dissolved P in runoff by approximately 66% under simulated rainfall, but also showed the smallest soil test P levels after 3 years of applications to corn.

Livestock Feed Management

Geographical areas with intense livestock production often import more nutrients in the form of feed than is exported in livestock or crop products. Necessarily, P will increase in the soils of such areas unless manure is exported. P inputs not only include the natural content of feed, but mineral supplements. Careful balancing of livestock rations may allow reductions in added P, reducing the P content of manure. Mahan and Howes (1995) estimate that balancing supplemental P to dietary intake requirements could reduce P use by 15%. Morse et al. (1992) recorded a 17% reduction in P excretion by dairy cows when daily P intake was reduced from 82 to 60 g/day. Maguire et al. (2005)

summarized the approaches for reducing dietary P in different species, and review the literature on the impact of these strategies on P forms in manures, and on P runoff from manure-amended soils.

Nitrogen levels in manure are also affected by livestock rations. For example, feeding dairy cattle a diet high in degradable intake protein caused more N to be excreted than a diet lower in degradable intake protein (Van Horn 1992). Crude protein levels in swine diets can also be decreased to minimize N losses from manure. Nitrogen losses during excretion, storage and field application decreased significantly when the amino acid balance in a swine diet was improved (Portejoie et al., 2004). Minimizing feed wastage and phase feeding (optimizing the diet to meet the needs of each particular swine age) are other ways to reduce N losses.

Phytate and phytase. Much of the P in corn is present in the form of phytic acid phytate. This form of P is unavailable to monogastric animals such as swine and poultry, with most being excreted in manure. Swine and poultry lack the phytase enzyme necessary to cleave the P from phytic acid. Ruminant animals, however, can fully utilize phytate-P because rumen microbes produce phytase. Only 10 to 20% of the total P in corn is available to swine and poultry, so that rations are usually supplemented with additional P.

Recently developed low-phytate corn (Raboy et al., 2000; Ertl et al. 1998) and barley (Dorsch et al., 2003) varieties have much higher available P content. Ertl et al. (1998) chemically induced mutations in corn, with one line having a 65% reduction in phytic acid with an equivalent increase in inorganic P. This line was backcrossed with other corn lines to produce hybrids with the low-phytate trait. In poultry and swine feeding trials, low phytate grain resulted in greater P availability for the animal and reduced P content in manure (Spencer et al., 2000; Veum et al., 2002; Thacker et al., 2003; Gollany et al. (2003)). Leytem et al. (2004) found that low-phytate barley grains fed to swine reduced total manure P concentrations and that manure P was not more soluble than the P originating from normal barley diets. Similar results were observed by Gollany et al. (2003), who found 42% lower total P content in manure originating from swine fed low-phytate corn compared to a normal corn diet, but both manures had similar phosphorus availabilities when applied to soil at the same P rates.

Supplementing feed with the phytase enzyme has decreased the need for supplemental P in livestock rations and reduced the P content of manure (Leytem et al., 2004; Jongbloed et al., 1992; Jongbloed et al., 2000; Bailleul et al., 2001). Phytase addition to animal diets has shown to improve P digestibility in more than 50% (Mroz et al., 1994; Lei et al., 1993; Holden and Tidman 1999; Roberson, 1999; Young et al., 1993; Adeola, 1995). The impact of phytase additions on availability of P in manure is being investigated to determine if reductions of total P in manure also reduce available P, the form of P of greatest environmental concern. Leytem et al. (2004) found that P in swine manure is mainly found in its inorganic form due to a complete hydrolysis of phytate in the large intestine of pigs and it is therefore highly soluble in water.

Pasture and Hayland Management

P losses from untilled pastures and hayland can be significant, especially when manure and fertilizer are applied to the soil surface, with most of losses being in the highly reactive dissolved form. The use of shallow tillage tools, such as harrows or "aerators", may increase infiltration, reducing P runoff losses.

Height of hay cutting can affect runoff loss potential. Zemenchik et al. (2002) measured P runoff from alfalfa-smooth bromegrass, smooth bromegrass, and alfalfa sods with rainfall simulation. There were no differences between the types of forage, but timing of rainfall relative to hay cutting had a large impact on P losses. Bioavailable P losses were 0.06 lb/ac (0.07 kg/ha) when rainfall occurred 6 weeks after harvest, 0.31 lb/ac (0.35 kg/ha) with rainfall 4 weeks after harvest, and 0.37 lb/ac (0.41 kg/ha) with rainfall immediately after harvest. Thus, leaving a greater height of uncut forage should be effective in reducing P losses from hayland.

Broadcast applications of maintenance P are often needed for forage legumes. If needed, applications should be made after the first cutting when runoff losses are lower. Split applications used for high yielding forages should be broadcast after the first and third cutting, but not in late fall or early spring (Schulte and Bundy 1988).

Management practices such as rotational grazing and exclusion of livestock from streams can reduce runoff. If livestock congregate in areas near streams, soils are compacted, increasing runoff where manure concentrations are greatest. Overgrazing eliminates vegetative cover that can slow runoff and trap P-bearing sediment.

Summary of N and P BMPs

Nitrogen

Conservation Tillage and Other Conservation Practices

- Conservation tillage reduces total N losses due to reduced sediment loss.
- Runoff and leaching losses of nitrate are not consistently affected by conservation tillage.
- Other conservation practices such as terraces and contouring reduce total N losses due to erosion reduction.

Nitrogen Rates

- Use a reasonable method to determine expected yields.
- Take credit for N applied as manure, in irrigation water, and fixed by legumes in rotation.
- Use appropriate soil tests to determine residual N.

Soil Testing

- Preplant soil tests are useful in drier climates.
- In areas where significant spring nitrate losses may occur due to leaching and/or denitrification, late spring or pre-sidedress N tests can determine if and how much additional N is needed.
- New soil test procedures, such as amino sugar tests, may be available in the future.
- Post harvest soil tests can determine if N management the previous season was adequate.

Application Precision

- New designs of manifolds may increase the uniformity of anhydrous ammonia distribution across applicators.
- Experimental applicators, such as injectors which form a compacted soil layer and surface ridge, may reduce N losses in the future.
- Variable rate applicators, combined with intensive soil or crop sampling, can allow correct N rates where fields vary in available N.

Irrigation Management

- Careful scheduling of irrigation based on soil moisture estimates and daily crop needs improves irrigation efficiency and reduces nitrate leaching.
- Sprinkler irrigation systems generally apply water more uniformly and in lower amounts than furrow irrigation systems, reducing nitrate leaching losses.
- Furrow irrigation efficiency can be improved by adjusting set time, stream size, furrow length, watering every other row, or the use of surge valves.

- Running irrigation water through every other furrow and applying N fertilizer in the nonirrigated furrow reduces nitrate leaching losses.
- Application of N fertilizer through irrigation systems facilitates supplying N when crop demands are greatest.
- PAM treatment with furrow irrigation reduces sediment and total N losses.

Crop Testing

- Leaf tissue tests can identify N deficiencies.
- Variations in chlorophyll content are being evaluated as a potential tool to facilitate variable rate N applications in-season.
- Post-black-layer corn stalk nitrate tests help to determine if N rates were low, optimal, or excessive, so that management changes can be made in following years.

Timing of N Applications

- Applying N sources close to when crops can utilize N reduces N loss risk.
- Sidedress application, usually made 4 to 6 weeks after planting crops, provides N just prior to the time of most rapid N uptake, and reduces risk of leaching and denitrification losses.
- Split applications, involving preplant and sidedress applications, allow efficient use of applied N and reduce risk of yield reductions, should sidedress applications be delayed.
- While fall N application may be discouraged in some areas, if anhydrous ammonia is applied in fall, wait until soil temperatures are below 50º F.

Fertilizer Application Method and Placement

- Injection or incorporation of urea or N solutions reduces volatility losses.
- In ridged crops, placing N fertilizers in a band in ridges makes N less susceptible to leaching.

Inhibitors

- Nitrification inhibitors maintain applied anhydrous ammonia in the ammonium form longer, reducing leaching and denitrification losses.
- Where fall N applications are appropriate, nitrification inhibitors reduce risk of leaching loss.
- Urease inhibitors temporarily block the function of the urease enzyme, maintaining urea-based fertilizers in the non-volatile urea form, and may reduce losses when these fertilizers are surface applied in high residue, conservation tillage systems.

Crop Rotation

- Legumes and other crops not needing supplemental N can utilize N remaining in the soil from previous N-fertilized crops, reducing nitrate leaching risk.

- Alfalfa can remove nitrate from the soil below the rooting depth of most annual crops.

Cover Crops

- Cover crops growing between the time annual crops are harvested and when successive crops are planted can scavenge N and other nutrients and prevent leaching.

Altered Drainage Tile Design

- Controlled drainage-subirrigation systems recycle nitrate leaching from the soil profile and reduce nitrate lost in tile drainage.
- Research is underway to alter tile installation to favor denitrification before tiles intercept drainage water.

Conservation Buffers

- Buffers trap sediment containing ammonia and organic N.
- Nitrate in subsurface flow is reduced through denitrification enhanced by organic energy sources placed in the subsoil by buffer plants.
- Buffer plants take up nitrate and other nutrients, preventing loss to water.

Constructed Wetlands

- Drainage tiles carry nitrate directly to surface water, bypassing potential processing by buffers. Constructed wetlands placed to process tile effluent reduce nitrate loads to surface water through denitrification.

Pest Management

- Proper pest management allows crops to attain their potential yields, utilizing applied N and reducing the amount of excess N available to loss.
- Bt corn prevents European corn borer feeding and associated stalk rots, which can cause corn to die early and leave excess N in the soil.

Breeding Crops for Efficient N Uptake

- Certain crop varieties may be able to more efficiently extract N from the soil.

Phosphorus

Conservation Tillage and Other Conservation Practices

- Conservation tillage consistently reduces runoff losses of total P.
- Runoff losses of dissolved P can be higher with no-till if fertilizer or manure are surface applied. Incorporating or injecting P sources below the soil surface reduces total and soluble P losses in all systems.
- Other soil conservation practices such as terraces and contour planting reduce total P losses due to reduced erosion.

Phosphorus Rates

- Use soil tests to determine agronomic rates.

- P rates may need to be reduced for environmental reasons in high risk areas.
- Environmental P thresholds are being developed by states to determine P rates protective of water resources.
- Consider P content of manure rather than solely N content.

Application Timing

- Avoid fertilizer and manure application to frozen soil.
- Tillage can incorporate fall-applied P fertilizers and manure, reducing runoff.
- Avoid fertilizer and manure applications to wet soils.

Fertilizer and Manure Placement

- Incorporation or injection of P sources reduces runoff losses.
- Starter applications can usually supply all of the maintenance P requirements for row crops.
- Variable rate application combined with intensive soil sampling can reduce fertilizer inputs and avoid overfertilizing high-testing areas of fields.

Irrigation Management

- Surface crop residue in furrow irrigated crops reduces sediment and P losses.
- PAM applied with furrow irrigation greatly reduces sediment, P, and organic material losses.

Conservation Buffers

- Buffers trap sediment and adsorbed P.
- Buffers need to be constructed and maintained to encourage sheet flow of runoff over the buffer.
- As sediment accumulates in buffers, changing their profile, sediment should be removed or buffers reshaped.
- Select buffer species based on adaptation to local conditions and other desired benefits of buffers such as wildlife habitat.

Cover Crops

- Cover crops reduce erosion between when annual crops are harvested and the following crop covers the ground, thus reducing runoff of sediment-adsorbed P.

Constructed Wetlands

- Constructed wetlands have most often been installed to encourage denitrification of nitrate. Their impact on P has been variable.

Phosphorus Index

- A coalition of U.S. scientists has developed a field-based planning tool that assesses the risk of P movement from soil to water. States are modifying the index to match local conditions.

Manure Management

- Determine nutrient content of manure to calculate appropriate application rates.
- Consider both P and N content of manure when determining rates.
- Fields testing low in P benefit most from manure applications.
- Consider risk factors such as nearness to streams, slope, presence of wells, sinkholes, surface tile inlets, and residences when selecting fields for manure applications.
- Incorporation or injection of manure reduces runoff risk.
- Avoid manure application to frozen soil.
- Using limited tillage to incorporate dry manure in one year over a several year period may allow the benefits of no-till production to be realized, while utilizing manure in an environmentally sustainable manner.
- Manure amendments such as alum reduce soluble P losses in runoff.

Livestock Feed Management

- Carefully balance livestock rations so that supplemental P is not excessive.
- Low-phytate corn varieties in nonruminant rations reduce the P content of manure.
- Supplementing nonruminant feed rations with the phytase enzyme reduces the need for supplemental P and reduces P content of manure.

Pasture and Hayland Management

- The use of shallow tillage tools such as harrows or "aerators" on pastures may increase infiltration and reduce runoff of manure or fertilizer.
- Rotational grazing reduces compaction, overgrazing, and nutrient runoff.
- Exclude livestock from access to streams.
- Make maintenance P fertilizer applications to forage legumes after the first cutting, when runoff losses are lower.
- Cutting hay higher to leave more stubble significantly reduces P runoff.

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